Optimising Water Quality and Impact Monitoring, Evaluation and Reporting Programs

Compiled by Jane Waterhouse

Australian Government

Department of the Environment,
Water, Heritage and the Arts

Reef & Rainforest RESEARCH CENTRE

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coasts | climate | oceans

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Acronyms and Abbreviations

ACTFR .... Australian Centre for Tropical Freshwater Research
AgSIP .... Agriculture – State-level Infrastructure Project
APSIM .... Agricultural Production Systems Simulator
BMP .... Best Management Practices
BSES .... Bureau of Sugar Experiment Stations
C .... Carbon
CDOM .... Coloured Dissolved Organic Matter
CMSS .... Catchment Management Support System
COTS .... Crown-of-Thorns Starfish
CPUE .... Catch per Unit Effort
CRC .... Cooperative Research Centre
CSIRO .... Commonwealth Scientific and Industrial Research Organisation
DEEDI .... Queensland Department of Employment, Economic Development and Innovation
DEWHA .... Commonwealth Department of the Environment, Water, Heritage and the Arts
DIN .... Dissolved Inorganic Nitrogen
DIP .... Dissolved Inorganic Phosphorus
DO .... Dissolved Oxygen
DON .... Dissolved Organic Nitrogen
DOP .... Dissolved Organic Phosphorus
DPC .... Queensland Department of the Premier and Cabinet
DPI&F .... Queensland Department of Primary Industries and Fisheries (now DEEDI)
EHMP .... Ecosystem Health Monitoring Program
EMC .... Event Mean Concentration
EnTox .... National Research Centre for Environmental Toxicology
EPA .... Environmental Protection Agency
EOC .... End of Catchment
FORAM Index .... Foraminifera in Reef Assessment and Monitoring Index
GAM .... Generalised Additive Model
GBR .... Great Barrier Reef
GBRMPA .... Great Barrier Reef Marine Park Authority
GPS .... Global Positioning System
IRCF .... Integrated Report Card Framework
IS .... Indicator Suite
LRE .... Loads Regression Estimator
M&E .... Monitoring and Evaluation
MERI .... Monitoring, Evaluation, Reporting and Improvement (Program)
ML .... Megalitres
MMP .... Reef Rescue Marine Monitoring Program
MODIS .... Moderate Resolution Imaging Spectroradiometer
Mt .... Megatonnes

MTSRF	Marine and Tropical Sciences Research Facility
N	Nitrogen
NRM	Natural Resource Management
P	Phosphorus
PN	Particulate Nitrogen
POM	Particulate Organic Matter
PP	Particulate Phosphorus
Pr	Primary
RCT	Resource Condition and Trend
RRRC	Reef & Rainforest Research Centre Ltd
Se	Secondary
SE	Standard Error
SEAP	Stream and Estuary Assessment Program
SIA	Stable Isotope Analysis
SIGNAL Index	Stream Invertebrate Grade Number – Average Level Index
SLATS	Statewide Landcover and Trees Study (Queensland)
SMART	Specific, Measurable, Achievable, Relevant and Timed
SRDC	Sugar Research and Development Corporation
SS	Suspended Sediment
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Sediment
UQ	The University of Queensland
WaterCAST	Water and Contaminant Analysis and Simulation Tool
WQIP	Water Quality Improvement Plan
WWF	World Wide Fund for Nature

Acknowledgements

About This Report

This report provides an overview of the key findings of research conducted through the Australian Government's Marine and Tropical Sciences Research Facility (MTSRF) relevant to water quality monitoring and evaluation in the Great Barrier Reef (GBR). It outlines the key constraints to water quality monitoring and evaluation in the GBR, advances in target setting methods and applications, and describes progress of monitoring and evaluation techniques to support the Reef Water Quality Protection Plan[1] and Reef Rescue[2] performance assessment in terms of indicator development and implementation. Information is also drawn from the Catchment to Reef Joint Research Program[3], funded as part of the CRC Reef Research Centre and the Rainforest CRC as precursors to the MTSRF, from 2002 to 2005, which aimed to develop appropriate monitoring methods for water quality and ecosystem health in aquatic ecosystems in the Wet Tropics and GBR World Heritage Area. Its goal was to provide a sound scientific basis for the development of monitoring tools, protocols and guidelines appropriate to the Wet Tropics. Many of the results presented in this report originated from this research, and have been further developed over the last four years through the MTSRF in conjunction with the Reef Rescue Marine Monitoring Program for water quality and ecosystem health monitoring in the GBR. While the research findings are also applicable elsewhere, particularly in tropical reef ecosystems, many of the more general outcomes have broader applications in environmental monitoring and evaluation programs.

A key achievement of the MTSRF has been the strong cooperation and collaboration between research institutions in project development and implementation. Many of the findings presented in this report were derived from large collaborative projects with funding from several sources in addition to the MTSRF, and the research institutions have also contributed significant in-kind resources. Supporting information derived from sources beyond the MTSRF is included in this report where necessary to provide context or to complete the discussion. Publications specifically generated under the MTSRF are identified in the reference list.

This report is one product in a series of information products that summarise MTSRF research findings relevant to managing water quality in the GBR. Other products include:

- A summary of MTSRF Water Quality Program highlights (Waterhouse and Devlin, 2010);
- 'Improved understanding of biophysical and socio-economic connections between catchment and reef ecosystems (Wet and Dry Tropics case studies), compiled by M. Devlin and J. Waterhouse (referred to as 'Catchment to Reef Connections' in this report) (Devlin and Waterhouse, 2010);
- 'Identification of priority pollutants and priority areas in Great Barrier Reef catchments', compiled by J. Waterhouse and J. Brodie (referred to as 'Priority Pollutants in the GBR' in this report) (Waterhouse and Brodie, 2010);
- A comprehensive review of pesticides in the Great Barrier Reef, compiled by K. Martin (Martin, 2010); and
- A synthesis of water quality and climate change interactions, and socio-economic influences on water quality management in the Great Barrier Reef, compiled by K. Martin (Martin, in prep.).

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[1] http://www.reefplan.qld.gov.au/
[2] http://www.nrm.gov.au/funding/2008/reef-rescue.html
[3] http://www.rrrc.org.au/catchment-to-reef/index.html

Executive Summary

The Great Barrier Reef (GBR) is a diverse ecosystem, which is bounded on its western side by a large number of large and small catchments. Protecting GBR ecosystems and the quality of the water they rely upon has become a major priority for resource managers and the community as a whole. Water quality and ecosystem health monitoring is needed to assess current status, identify existing and emerging problems, evaluate the consequences of various anthropogenic land and water use practices, devise improved practices and assess the effectiveness of management measures.

The Australian Government's MTSRF has generated significant outcomes for informing the design and implementation of GBR water quality monitoring, evaluation and reporting programs. In particular, a monitoring and evaluation framework that incorporates biophysical, social and economic aspects of the system at multiple scales has been developed. This framework includes a range of monitoring and modelling activities to combine system attributes at several scales from plot/paddock, to sub catchment, catchment and regional scales and, ultimately, across the entire GBR. Suitable indicators for measuring ecosystem status and response have been developed and tested for the GBR and the associated catchments. The indicators incorporate all aspects of the system that managers need to understand to assess the performance of actions in the catchment and the response in the GBR, including measures of management practice status and change, catchment health, catchment loads, estuarine health, marine water quality and marine ecosystem health. This suite of indicators is summarised in Figure i. In some cases, thresholds for these indicators are established, which form the basis for the definition of guidelines to trigger a management response. The best ways to report indicators have also been considered.

Importantly, some of the constraints that previously hampered GBR water quality monitoring and evaluation programs have now been overcome, thanks to MTSRF-funded research. Specifically, best estimates of current contaminant loads to the GBR have been generated (Kroon *et al.,* 2010; Brodie *et al.,* 2009a), techniques for improved estimation of loads have been developed (e.g. Wallace *et al.,* 2009a, 2010b; Kuhnert *et al.*, 2009, 2008; Kuhnert and Henderson, 2010; Wang *et al.*, 2009; Lewis *et al.*, 2009a, 2007a), more efficient and robust indicators have been developed and tested for freshwater (Pearson *et al.*, 2010a), estuarine (Sheaves *et al.*, 2010) and marine ecosystems (Cooper *et al.*, 2009; Fabricius *et al.*, 2010a, 2010b), indicators of social and economic status and governance arrangements are being developed (van Grieken *et al.*, 2010a; Lynam *et al.*, 2010a; Taylor and Robinson, 2010) and receiving water models are being established (Brinkman *et al.*, 2010; Maughan and Brodie, 2009). In addition, thresholds of concern for priority pollutants have been established for marine ecosystems (GBRMPA, 2009; De'ath and Fabricius, 2010) and used as the basis for the Great Barrier Reef Water Quality Guidelines (GBRMPA, 2009). Pollutant thresholds for freshwater and wetland ecosystems have also been investigated (Pearson *et al.*, 2010a; Wallace *et al.*, 2010a). This report provides an overview of these outcomes, with particular emphasis on aspects that have and can be applied in GBR monitoring programs, as well as other circumstances in national and international situations.

Outcomes of this research are already being taken up and used by managers to improve the effectiveness of water quality monitoring programs for the GBR. For example, the multi-scale, multi-disciplinary 'Paddock to Reef'[4] monitoring and modelling framework has been used to inform the development of the Integrated Monitoring, Modelling and Reporting Program to support the evaluation of the Reef Plan and Reef Rescue initiatives, and researchers continue to develop improved monitoring and evaluation techniques and

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[4] http://www.reefplan.qld.gov.au/publications/paddock-to-reef.shtm

indicators for continued refinement of program design. Many of the coral and seagrass indicators developed and tested through the MTSRF are already operational as part of the Reef Plan / Reef Rescue Marine Monitoring Program.

The whole-of-system monitoring approach was also used in the development of regional water quality plans, including the WQIPs for the Tully, Barron, Townsville-Thuringowa (Black Ross), Mackay-Whitsunday and Burnett Mary regions. These programs assisted in the identification of priority contaminants and priority areas for each region. In conjunction with revised and improved pollutant load estimations, the findings have informed the prioritisation of Reef Rescue expenditure in these regions. The findings of the catchment and instream health research can be used to assess the condition of Wet Tropics streams and wetlands, which is of interest to the Queensland Government and regional natural resource management groups.

Finally, many of the most significant influences of the research on management decisions have been through the participation of MTSRF researchers in steering committees and technical groups coordinated by management agencies. MTSRF researchers are able to contribute their knowledge and synthesis of the research findings directly into the management processes; in many cases their contribution to discussion instigates interest which is subsequently supported through the provision of written evidence. Examples of these activities include the range of technical groups and forums coordinated for the regional WQIPs and revision of the Reef Plan (DPC, 2009), design workshops for the Paddock to Reef Program and ongoing participation in the associated Technical Advisory Group, the expert workshops convened for the multi-criteria analysis for prioritising Reef Rescue investment, participation in various committees for the Queensland Wetland Program, and involvement in several research prioritisation workshops which have informed the Reef Plan and Reef Rescue Research and Development Strategies. Knowledge gained through the MTSRF and other research also contributed to the 2008 Scientific Consensus Statement for Water Quality in the Great Barrier Reef (Brodie *et al.*, 2008a).

MTSRF-funded research has also revealed knowledge gaps and new areas of research that should be progressed to inform continuous improvement of monitoring and evaluation programs, both in the GBR and elsewhere. Future research directions are summarised for each system component that has been studied through the MTSRF. Progression of the future research directions highlighted in this report will assist managers of GBR water quality to further improve the design of monitoring and evaluation programs within an adaptive framework. Continued alignment of monitoring programs with research programs, as has been the case for the Reef Plan / Reef Rescue Marine Monitoring Program for several years, will assist in this process.

Management Practices

Management practices
- **Types of classes / management practices**
- **Adoption of practices (by class)**
- **Plot/paddock water quality – suspended sediments, nutrients, pesticides**
- **Cost effectiveness of practices**
- Capacity to adopt (landholder profiling)

Catchment health

Catchment health
- Habitat variables inc riparian extent & condition, flow, aquatic plants
- Physical condition inc velocity, bank stability, channel form, width, depth, sediment
- Water quality
- Invertebrate species & family richness
- Fish species richness & assemblage composition
- Abundance & diversity alien fish
- Floodplain lagoon & wetland benthic habitat
- **Wetland & riparian extent**

Catchment loads

Catchment loads
- **Sub-catchment & catchment water quality – suspended sediments, nutrients, pesticides**
- **Deterministic process-based model**
- **Statistical modelling framework - Loads Regression Estimator**

Socio economic influences

Socio economic influences
- Examples include:
- Population pressures
- Quality of life
- Public/private benefit of practice change
- **Use indicators e.g. No. guest nights, reef visits**
- General capacities & participation
- Evaluation of management performance
- Networks & links to other organisations

Estuarine health

Estuarine health
- **Ambient water quality – dissolved oxygen, pH, temperature, nutrients, suspended sediments**
- Change in fish assemblage taxonomic structure
- Fish assemblage indices
- Fish Stable Isotope Analysis ($$)
- Fish Biomarker Analysis ($$)

Marine Water Quality

Marine water quality
- **Ambient – suspended sediments, nutrients, carbon, salinity, temperature, chlorophyll, pesticides**
- **Flood plume – suspended sediments, nutrients, salinity, temp, chlorophyll, pesticides**
- **Flood plume extent & exposure**

Coral Health

Coral health
- *Porites* brightness
- Macro-bioeroder density in massive living *Porites*
- **Foram index**
- **Macroalgal cover**
- *Acropora* **cover**
- *Acropora* **/ Hard coral cover ratio**
- **Hard coral juvenile density & richness**
- **Soft coral juvenile density & richness**

Seagrass Health

Seagrass health
- **Tissue nutrient ratios (N:P, C:N, C:P)**
- **Seed bank & reproductive structure surveys**
- **Leaf number per shoot & area**
- **Seagrass species composition**
- **Changes in seagrass cover & meadow area**

Figure i. Summary of the recommended suite of indicators for monitoring water quality and ecosystem health in the Great Barrier Reef and its catchments, developed and tested through the MTSRF. Indicators highlighted in bold have already been adopted by the Reef Plan Paddock to Reef Integrated Monitoring, Modelling and Reporting Program.

1. Introduction

Tropical Queensland hosts a unique and diverse assemblage of interdependent aquatic ecosystems and includes most of the catchment area draining into the Great Barrier Reef (GBR) lagoon, as well as the GBR ecosystem itself. Protecting these ecosystems and the quality of the water they rely upon has become a major priority for resource managers and the community as a whole. Water quality and ecosystem health monitoring is needed to assess current status, identify existing and emerging problems, evaluate the consequences of various anthropogenic land and water use practices, devise improved practices and assess the effectiveness of management measures. This needs to be done at a range of spatial scales commensurate with different scales of management from individual properties and/or water bodies through sub-catchment and river reaches, to the whole GBR catchment (Arthington and Pearson, 2007; Brodie *et al.*, 2007a).

The GBR system is bounded on its western side by a number of large and small catchments. Rivers discharging from these catchments transport considerable amounts of suspended sediments, nutrients and pesticide residues into the GBR lagoon (Brodie *et al.*, 2001; Furnas, 2003). The loads of these substances have increased greatly over the last two hundred years as agricultural and other development has proceeded (Kroon *et al.*, 2010): for example, suspended sediment loads have increased by a factor of 5.5, total nitrogen (TN) by a factor of 5.8 (consisting of dissolved inorganic nitrogen (DIN) (3x), dissolved organic nitrogen (DON) (2x) and particulate nitrogen (PN) (~70x)), and total phosphorus (TP) by a factor of 9 (consisting of dissolved inorganic phosphorus (DIP) (~5x), dissolved organic phosphorus (DOP) (~2x) and particulate phosphorus (PP) (~15x)). Pesticide residues are now detectable in the GBR lagoon year-round (Lewis *et al.*, 2009b). The increase in these loads of contaminants is believed to have had considerable negative effects on ecosystems of the GBR region, both fresh (Arthington *et al.*, 1997; Wallace *et al.*, 2009a) and marine waters (Brodie *et al.*, 2001, 2005; Fabricius, 2005; Fabricius *et al.*, 2005; De'ath and Fabricius, 2010).

In an attempt to reduce contaminant load to the GBR, the Australian and Queensland Governments introduced the Reef Water Quality Protection Plan (Reef Plan) in 2003 (DPC, 2003). Until 2008, one of the primary implementation mechanisms for Reef Plan was through Water Quality Improvement Plans (WQIPs) developed for specific catchment areas within Regional Natural Resource Management (NRM) regions across the GBR catchment. WQIPs have been developed for the Douglas Shire area and Tully-Murray basin in the Wet Tropics, the Burdekin basin, Ross-Black basin (Townsville-Thuringowa), the Mackay-Whitsunday region and the Burnett basin. A regional water quality plan was also established for the Fitzroy NRM Region. The development of each WQIP required the identification of key water quality issues in the relevant region and definition of targets to protect important assets with emphasis on the GBR World Heritage Area. These targets reflect the community desire to maintain a healthy GBR and relied on GBR water quality guideline trigger values for key pollutants (GBRMPA, 2009). A component of this process in each case included a water quality monitoring program to identify issues and quantify sources.

In late 2007, the Australian Government committed A$200 million over five years for a Reef Rescue program 'to tackle climate change and improve water quality in the Great Barrier Reef' (Australian Government, 2007). This package included substantial funding (A$146 million) for a Water Quality Grants Scheme (for improved land management practices), and supporting monitoring, reporting and research programs, with additional funding to build partnerships. This program is building heavily on the planning for management developed through the WQIPs in each region.

In 2009, the Queensland and Australian Governments committed to the implementation of a comprehensive monitoring, modelling and reporting program to assess the progress of the Reef Plan and Reef Rescue initiatives. The program design uses a combination of monitoring and modelling techniques to inform progress on achieving various water quality related targets by 2013 (see Section 3). The program, referred to as the Reef Plan Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (herein referred to as the Paddock to Reef Program; see DPC, 2010), incorporates activities across geographic regions and scales – from plot to paddock to farm/multi-farm scale to sub-catchment and catchment scale, to coastal and inshore reef environments. It involves monitoring and modelling a range of attributes, including management practices and water quality at the paddock, sub-catchment, catchment and marine scales to report against Reef Plan and Reef Rescue targets. This approach requires the ability to link the monitoring and modelling outputs at each scale and then across scales, and is described in further detail in Bainbridge *et al.* (2009a). The program engages at least eighteen organisations including Queensland and Australian government agencies, regional natural resource management bodies, research institutions and industry groups. Many of the concepts underpinning the program design were developed through the MTSRF, as supported by the information presented further in this report.

While the management of water quality for improvement of GBR ecosystem health has been the primary focus of government policy and investment in recent years, catchment waterway and instream health is also an important priority for natural resource management, particularly for regional communities and NRM bodies, and there are established linkages between instream and catchment health and downstream impacts (Arthington *et al.*, 1997; Arthington and Pearson, 2007). The importance of the catchments to reef health was recognised in the Reef Plan, although there is still a belief that achieving end-of-river targets will be the answer to reef health. As identified in Brodie *et al.* (2009b) and Pearson *et al.* (2010a) this view misses some important points, for example:

- The end of the river does not represent the entirety of the catchment, as many of the floodplain discharges are separate from the main river and the measure of 'management change' is therefore inaccurate;
- The influence of chronic delivery of contaminants during the non-flood period is largely unknown;
- Good end-of-river water quality may be achievable despite poor habitat and water quality in the catchment (e.g. invasive weeds or dams can arrest contaminant transport); and
- Freshwater and marine systems are part of a continuum, especially for elements of the biota that must use both (e.g. barramundi), and so require good conditions throughout the system.

These relationships are discussed in more detail in the accompanying report, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.). The development of indicators and thresholds of concern for freshwater ecosystems, including coastal wetlands, has been an important part of the MTSRF water quality research program, and is discussed in more detail in Section 4.3.

This report provides an overview of the key findings of MTSRF-funded research relevant to water quality monitoring and evaluation in the GBR. It outlines the key constraints to water quality monitoring and evaluation in the GBR, advances in target setting methods and applications, and describes progress of monitoring and evaluation techniques to support the Reef Plan and Reef Rescue performance assessment in terms of indicator development and implementation. Information is also drawn from the Catchment to Reef Joint Research Program, funded as part of the CRC Reef Research Centre and the Rainforest CRC as

precursors to the MTSRF from 2002 to 2005, aimed to develop appropriate monitoring methods for water quality and ecosystem health in aquatic ecosystems in the Wet Tropics and GBR World Heritage Areas. Its goal was to provide a sound scientific basis for the development of monitoring tools, protocols and guidelines appropriate to the Wet Tropics. Many of the results presented in this report originated from this research, and have been developed over the last four years through the MTSRF in conjunction with the Reef Rescue Marine Monitoring Program for water quality and ecosystem health monitoring in the GBR.

2. Constraints to GBR water quality monitoring and evaluation

Challenges in measurement and evaluation of water quality outcomes, and the transport, fate and impact of contaminants in the GBR are related to geographic scales, temporal variability, system noise and time lags in the system, and uncertainties in system understanding (Haynes *et al.*, 2007; Bainbridge *et al.*, 2009a). There are also a range of institutional and governance constraints. All of these constraints are also reflected in the capacity to set measurable and achievable performance targets.

A major limitation in detecting improvements in management practices and measurable outcomes in GBR ecosystem health is the ability to detect the signal of change in the system. Noise in the signal is due to system variability (influenced by rainfall, flow, topography), natural occurrence of sediments and nutrients in the system and limitations of the capacity to monitor and model material transport and fate (Waterhouse *et al.*, 2009).

Time lags in the system can have a marked effect on the ability to measure change in the system as a result of management response. These time lags can be associated with material transport within catchments and into the GBR lagoon, and between catchment management actions and the resultant changes in water quality at varying downstream catchment scales. In addition, time lags will vary depending on what water quality parameter is being measured. For instance, sediment lag times may be much longer than reductions in dissolved inorganic nitrogen or pesticide concentrations in waterways, where management actions to reduce these parameters (e.g. optimisation of herbicide application through the use of new technologies such as shielded sprayers) may result in reductions in concentrations within a period of months to 2-3 years (as shown in Table 1). It is critical to acknowledge that the first detectable changes towards water quality improvement will be attributed to management practice change and the longer-term response will relate to load outputs. This highlights the importance of innovative monitoring and modelling techniques, and an improved understanding of system dynamics to inform management decisions relating to water quality management in the GBR.

While the holistic catchment view is important in understanding large-scale processes and management issues for freshwater and instream health (e.g. flow regimes, hydrological connectivity, end-of-river outputs), it is at smaller scales that many ambient processes need to be understood (e.g. oxygen dynamics) along with biological responses to thresholds of concern. There is a need for considerable research effort and regionally specific knowledge to understand, monitor and evaluate catchment health at the GBR-wide scale. MTSRF-funded research has sought to understand some of these relationships to guide future management activities and, in some cases, monitoring and evaluation programs specifically. Current understanding of catchment ecosystem health and function is described in more detail in the report 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.), and the key advances in techniques to inform monitoring program design are described below in Section 4.

The primary constraints to freshwater and instream health monitoring currently include:

- Lack of a long-term coordinated freshwater and instream health monitoring program in the GBR catchments;
- Lack of coordinated, strategic long-term government and NRM investment;
- Limited knowledge of the ecology of freshwater ecosystems (gradually being addressed by MTSRF, the Catchment to Reef Joint Research Program, and other programs);

- Limited understanding of cause-effect relationships between climate, water/habitat quality, animal and plant species and communities, and ecosystem processes;
- Limited understanding of the likely effects of changed on-farm management practices and consequences for instream water quality and ecosystem condition;
- Uncertainty about thresholds and appropriate targets for the long term;
- Little cognisance of the importance of the catchment and its waterways in their own right, as a centre of unique biodiversity, as well as a source and deliverer of contaminants to the GBR; and
- Limited understanding of the importance of connectivity between freshwater, estuarine and marine systems for maintenance of biodiversity and ecosystem processes.

Current understanding of the cause-effect relationships between declining water quality and GBR ecosystem response has progressed significantly in recent years, particularly through the work of Pearson, Wallace and others in wetland connectivity and freshwater ecosystems (particularly in the Wet Tropics) (for example see Pearson *et al.*, 2010a, 2010b; Wallace *et al.*, 2010b, 2007; Karim *et al.*, submitted, 2010a, 2010b, 2009a, 2009b; Godfrey, 2009; Arthington and Pearson, 2007; Mackay *et al.*, 2010), and Fabricius and others in marine ecosystems (for example see Fabricius and De'ath, 2004; De'ath and Fabricius, 2010). This has allowed development of quantitative indices of wetland connectivity and more effective indicators of ecosystem pressures and response, contributing to improved design of water quality related monitoring programs in the GBR. Examples of these are detailed in Sections 4.3 and 4.5 of this report.

There are also a range of constraints associated with limitations of current institutional and governance arrangements. These include uncertainties in the estimation of pollutant loads and, therefore, performance; uncertainties in target values currently being set due to available scientific knowledge at the time, particularly a lack of long-term monitoring datasets; short funding cycles; timeframes required by government for delivery of Reef Rescue performance assessment (e.g. yearly reporting, short term targets of less than five years) and barriers to interagency collaboration such as those explored in Taylor and Robinson (2010), also discussed in further detail in Section 4.6.

Table 1. Timeframes for water quality trends/signals to be detected for three example parameters at varying spatial scales from paddock to reef as a result of management actions (from Bainbridge *et al.,* 2009a). Suspended sediment in the Burdekin Rangelands, dissolved inorganic nitrogen in the lower Burdekin, and herbicides on the Tully floodplain have been used as examples to demonstrate varying scales.

Management actions/ remedial activity	Water Quality Parameter		
	Suspended sediment	**Dissolved nitrogen**	**Herbicides**
	Erosion control mechanisms for grazing lands e.g. riparian fencing and wet season spelling	Reduction of fertiliser use in cropping lands e.g. implement *Six Easy Steps*	Minimise/optimise pesticide use through new technologies e.g. shielded sprayers, control traffic
Timeframe of water quality trends/signals being detected at different spatial scales			
Paddock/Plot Scale	Likely to be detected after 2-3 wet seasons e.g. Virginia Park Station	Months to 3 years; depends on the nitrogen stored in the system (e.g. soil, organic matter) e.g. BRIA paddock	Months to 1 year; depends on previous usage and residuals in the system. e.g. Tully paddock
Local Scale **e.g. immediate drainage line/ local waterway**	Likely to be detected within 5-10 years depending on system noise e.g. Weany Creek	Likely to be detected within 1-3 years; depending on rate of adoption within local area and system noise e.g. local cane drain	Likely to be detected <1 year due to relatively short half life (i.e. diuron half-life in soil is 90 days, and likely complete life <2 years) e.g. local cane drain
Sub-catchment Scale	Greater than 10 years; even for major scale land management interventions across the sub-catchment e.g. Fanning River	Expect to measure change <10 years if sugarcane is dominant land use in catchment and management change is widely adopted; particularly if detailed pre-monitoring data are available e.g. Upper Barratta Creek	Expect to measure change within 2 years if sugarcane is dominant land use in catchment and management change is widely adopted; particularly if detailed pre-monitoring data are available e.g. Davidson Creek
End-of-catchment Scale	Likely >50 years (major erosion control management intervention across the Burdekin); dilution of signal as only small % of total catchment area under improved management at any one time, and hydrological variability or noise is high. e.g. Burdekin River (Inkerman)	Expect to measure change <10 years if sugarcane is dominant land use in catchment and management change is widely adopted; particularly if detailed pre-monitoring data are available e.g. Barratta Creek (Bruce Hwy)	Change detected <2 years, however may be dilution effect depending on amount of cane in catchment, and proportion of uptake by the industry within this catchment e.g. Tully River (Euramo)
Estuarine and Marine Scale **e.g. coastal waters within adjacent bay**	Likely >50 years before change in turbidity; limited likelihood of detecting signal from this management action due to size of catchment. e.g. Upstart Bay	Likely to detect change in chlorophyll from this management action (major nitrogen fertiliser reduction across the lower Burdekin sugar lands) <20 years, with variability due to other sources of nutrients (e.g. Burdekin plume), seasonal variations in nitrogen cycling and sea water mixing. e.g. Bowling Green Bay	Changes likely to be detected within two years in the flood plume, however signal may be difficult to detect if the coastal waters are also influenced by larger river flood plumes (e.g. Herbert or Murray Rivers) e.g. Dunk Island and Family Island Group

2.1 Overcoming constraints through applied research

MTSRF-funded research has overcome some of the constraints for current monitoring and evaluation for water quality improvement in the GBR. For example, best estimates of current contaminant loads to the GBR have been generated (Kroon *et al.,* 2010; Brodie *et al.,* 2009a), techniques for improved estimation of loads have been developed (e.g. Wallace *et al.*, 2009a, 2010b; Kuhnert *et al.,* 2009, 2008; Kuhnert and Henderson, 2010; Wang *et al.*, 2009; Lewis *et al.*, 2007a), more efficient and robust indicators have been developed and tested for freshwater (Pearson *et al.*, 2010a), estuarine (Sheaves *et al.*, 2010) and marine ecosystems (Cooper *et al.*, 2009; Fabricius *et al.*, 2010a, 2010b), indicators of social and economic status and governance arrangements are being developed (van Grieken *et al.*, 2010a; Lynam *et al.*, 2010a; Taylor and Robinson, 2010) and receiving water models are being established (Brinkman *et al.*, 2010; Maughan and Brodie, 2009). In addition, thresholds of concern for priority pollutants have been established for marine ecosystems (GBRMPA, 2009; De'ath and Fabricius, 2010) and used as the basis for the Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009). Pollutant thresholds for freshwater and wetland ecosystems have also been investigated (Pearson *et al.*, 2010a; Wallace *et al.*, 2010a). This report provides an overview of these outcomes, with particular emphasis on aspects that have and can be applied in GBR monitoring programs, as well as transferability to other circumstances nationally and internationally.

3. Setting performance targets for monitoring and evaluation

Overall performance assessment of large-scale initiatives such as the Reef Plan and Reef Rescue programs require a multi-scaled approach to evaluation and reporting. Definition of targets and long-term goals for GBR ecosystems, and identification and implementation of management actions required to meet those targets in priority areas, has been the primary management approach adopted for managing water quality in the GBR. This approach is underpinned by the concept of an interconnected catchment and GBR system, where targets for implementation of land-use practices in sub-catchments influence water quality in outflowing streams. These become inputs affecting end-of-catchment water quality or loads, which in turn affect receiving water quality and ecosystem responses. There is a further nested spatial hierarchy, with multiple paddocks or land-use units within sub-catchments, multiple sub-catchments within catchments, and potentially multiple catchments affecting GBR sub-regions. To add to the complexity, there are multiple objectives underpinning many of the actions, and these need to be recognised as part of assessment and reporting.

The system includes relationships within and across catchments to the GBR, so that the linkages between catchment actions and GBR health, and within the components of the system (e.g. between water quality and coral health), need to be understood and quantified. Linkages between biophysical, social and economic dimensions of the system are also critical so that realistic targets and implementation strategies can be developed and assessed. Finally, linkages across scales are necessary so that the sum of catchment and regional activities can be assessed to determine whether the existing and proposed activities are sufficient to achieve the Reef Plan goals (Eberhard *et al.,* 2008). Techniques to link between scales and landscapes are therefore required. Accordingly, predictive models are required to 'scale up' the monitoring results. This concept, which is described further in Section 4.1, forms the basis of the design of the Paddock to Reef Program.

In accordance with this whole-of-system concept, one of the requirements of the Reef Plan and Reef Rescue initiatives is that performance targets are established at various scales that will protect valuable assets. At a GBR-wide scale, the recently revised Reef Plan (DPC, 2009) specifies management action, catchment condition and end-of-catchment pollutant load targets for 2013, reported by catchment, regional and GBR-wide scales (see Table 2). These are largely complementary to a set of targets designed to assess performance of the Reef Rescue investment and set a fifty percent reduction in nitrogen and phosphorus loads at end of catchments by 2013 and twenty percent reduction in sediment loads by 2020, in addition to several adoption targets for sugar cane and grazing management practice (Table 2). The Reef Rescue targets specify different reductions for dissolved and particulate nutrients (25% and 10% respectively) and sediment (10%) end-of-catchment loads to reflect management priorities. Catchment condition targets (groundcover extent, and wetland and riparian area extent and condition) and management practice adoption targets are also specified in the Reef Plan. The Reef Plan also includes an additional long-term goal to ensure that by 2020 the quality of water entering the GBR from adjacent catchments has no detrimental impact on the health and resilience of the Reef. Within GBR waters, water quality guidelines have been defined to sustain the health of GBR ecosystems including regionally specific and cross-shelf guidelines for nutrients and chlorophyll, suspended sediments and recommended trigger levels for a range of pesticides (GBRMPA, 2009). These targets and guidelines are necessary to justify the level of investment on the basis of a known 'required' level of pollutant reduction to meet the ecosystem requirements of the GBRWHA, and are described in further detail in Section 4.5.

Table 2. Reef Water Quality Protection Plan and Reef Rescue (shaded cells) targets for the Great Barrier Reef. EOC = End of Catchment.

Target	Scale (area) for reporting	Reporting frequency
50% Reduction in N load at EOC by 2013 Total N plus DIN, DON, PN	EOC for all GBR catchments	Annual
50% Reduction in P load at EOC by 2013 Total P plus DIP, DIP, PP	EOC for all GBR catchments	Annual
Reduce the load of dissolved nutrients from agricultural lands to the GBR lagoon by 25% by 2013	EOC for all GBR catchments	Annual
Reduce the discharge of particulate nutrients from agricultural lands to the GBR lagoon by 10% by 2013	EOC for all GBR catchments	Annual
50% Reduction in pesticide load at EOC by 2013	EOC for all GBR catchments	Annual
Reduce the load of chemicals from agricultural lands to the GBR lagoon by 25% by 2013	EOC for all GBR catchments	Annual
Minimum 50% late dry season groundcover in DT grazing lands by 2013	Sub catchment for Burdekin and Fitzroy	Annual
20% Reduction in sediment load by 2020	EOC for all GBR catchments	Annual
Reduce the discharge of sediment from agricultural lands to the GBR lagoon by 10% by 2013	EOC for all GBR catchments	Annual
No net loss or degradation of wetlands	Catchment for all GBR catchments	Yr 1 and 5
Condition and extent of riparian areas improved	Catchment for all GBR catchments	Yr 1 and 5
80% of landholders adopted improved practices	Sub catchment or catchment by sector	Annual
To increase the number of farmers who have adopted land management practices that will improve the quality of water reaching the reef lagoon by a further 1,300 over three years	Sub catchment or catchment by sector	Annual progress; major report 2011
50% of landholders adopted improved practices (grazing)	Sub catchment or catchment	Annual
To increase the number of pastoralists who have improved ground cover monitoring and management in areas where runoff from grazing is contributing significantly to sediment loads and a decline in the quality of water reaching the reef lagoon by a further 1,500 over three years	Sub catchment or catchment	Annual progress; major report 2011

Historically, although targets were set (e.g. Brodie *et al.*, 2001), the process was constrained by lack of adequate knowledge and was therefore somewhat ad hoc and lacking in scientific transparency. Approaches to establishing more meaningful targets have progressed substantially in recent years, predominantly through the WQIPs in each GBR Region. The current target-setting process attempts to provide more scientifically robust targets using linked models from the paddock scale, to sub catchment and end of catchment scale to the reef (Brodie *et al.*, 2009b). The process follows the principle of the SMART (Specific, Measurable, Achievable, Relevant and Timed) targets developed for the GBR catchments by McDonald and Roberts (2006), where water quality targets are explicitly determined from achievable land management practices and set within an adaptive management framework.

This process also allows analysis of management options by running and exploring scenarios, and can assess potential progress towards scientifically validated targets for various management options. The adopted target-setting process also aims to closely link end of catchment pollutant loads with marine ecosystem objectives, which greatly increases its complexity, but is known to be an important component to produce realistic targets (e.g. Borsuk *et al.*, 2004; Karr and Yoder, 2004).

Brodie and others (2009b) provide an overview of the target setting approach adopted in the Tully-Murray and Burdekin River catchments. The approach uses a combination of monitoring data and material transport models including the SedNet and ANNEX catchment models for estimating suspended sediment and nutrient end-of-catchment loads (e.g. Kinsey-Henderson *et al.*, 2007; Armour *et al.*, 2009). To make predictions about the link between these pollutant load estimates and marine ecosystem response, ChloroSim (Wooldridge *et al.* 2006), a combined hydrodynamic/chlorophyll-nitrate correlation model for the GBR (Wooldridge *et al.* 2006), was also applied in the Tully-Murray example. This type of model is currently not available in other regions. The model links a quantitative river discharge parameter (i.e. dissolved inorganic nitrogen (DIN) concentration in event flows) with a quantitative indicator of health in the marine environment (i.e. chlorophyll concentration). This relationship has been confirmed for the GBR north of the Burdekin River, where observed summer chlorophyll concentrations in the inner-shelf areas increase significantly with the export of elevated DIN from the adjacent river catchments (Wooldridge *et al.*, 2006).

In a highly dynamic system like the GBR, there is likely to be considerable uncertainty in estimates of progress against targets. Therefore, a transparent process of uncertainty analysis is important for stakeholders involved in target load setting processes (DePinto *et al.*, 2004), including all inputs and assumptions. However, both SedNet/ANNEX and ChloroSim are deterministic models and, while it is generally understood that these models have high degrees of uncertainty, quantitative uncertainty estimates are not part of either model's output. Uncertainty in water quality model predictions is inevitably high owing to model equation error, parameter error and boundary condition problems (McIntyre and Wheater, 2004). Moreover, errors leading to target uncertainty propagate through the chain of models we have used to set the targets. Some uncertainty analysis has been studied for the SedNet/ANNEX model in a general setting, focusing on inputs such as soil nutrient data (Sherman and Read, 2008), hydrology and other inputs (Newham *et al.*, 2003), erosion source inputs (Kuhnert *et al.*, 2010), vegetation cover and gully density assumptions (Herr and Kuhnert, 2007; Dougall *et al.*, 2007), and comparing model outputs to equivalent monitoring results (Bartley *et al.*, 2007; Sherman *et al.*, 2007; Armour *et al.*, 2007, 2009). Thus, for SedNet/ANNEX, it is possible to make some semi-quantitative estimates of model uncertainty for specific model runs. Continuing work by Kuhnert and others (see Kuhnert *et al.*, 2010) aims to develop techniques for estimating uncertainty in load estimates. No model uncertainty studies have been conducted for ChloroSim, and estimates of uncertainty for this model are based on an 'expert judgement' (qualitative assessment) approach.

While considerable effort has gone into setting water quality targets for the GBR, less attention has focused on the assessment of collaborative delivery options needed to meet these targets. Robinson and others (2009a) have adapted the SMART criteria to assess partnership needs to deliver voluntary water quality management programs proposed at a regional scale in the GBR catchments, described further in Section 4.6.2.

4. Optimising program design for GBR water quality monitoring and evaluation

Design and implementation of GBR water quality monitoring and evaluation programs must take all of the above constraints and issues into consideration. The following section summarises MTSRF research findings that are informing optimal program design. Additional information from sources outside the MTSRF is also incorporated to provide contextual information where required.

4.1 Adopting whole-of-system approach

The basic framework developed for the overall design for monitoring and evaluation of water quality in the GBR is shown in Figure 1. It involves monitoring and modelling a range of attributes including management practices and water quality at the paddock, sub catchment, catchment and marine scales. This approach, which requires the ability to link the monitoring and modelling outputs at each scale and then across scales, is described in further detail in Bainbridge *et al.* (2009a) (the information below has been extracted from this publication with permission from the authors). This framework has been adopted in many regional WQIPs, and forms the basis of the design of the Paddock to Reef Program.

Figure 1. Monitoring from paddock/plot to marine ecosystem scale. Management action targets, resource condition targets and GBR lagoon trigger values are also highlighted (from Bainbridge *et al.,* 2009a).
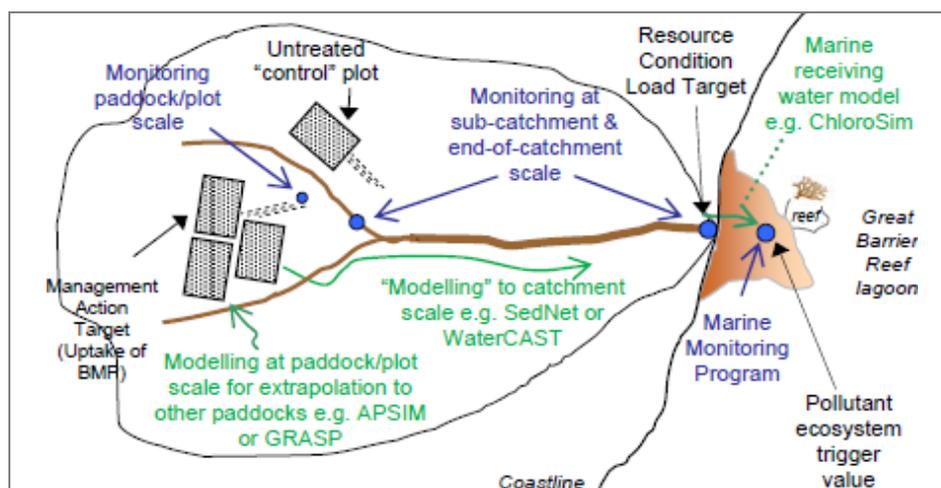
Plot/paddock-scale monitoring provides information on the unit degree of water quality improvement for specific management practices, for example reduction of dissolved inorganic nitrogen (DIN) loss from paddock in kg/ha after implementation of the fertiliser management program 'Six Easy Steps' (Schroeder *et al.*, 2005). Untreated 'control' paddock water quality data are also required to determine the water quality improvement resulting from the changed practice. **Plot/paddock models** such as APSIM, HowLeaky or GRASP are also used to generate this information. The extent of **management practice uptake** needs to be integrated with the knowledge generated through the plot/paddock-scale monitoring and modelling for upscaling from plot/paddock, sub-catchment and catchment scales.

The plot/paddock-scale monitoring and modelling data and management practice uptake information are then used as inputs into a **catchment model**. The catchment models are fundamental to describing how management impacts on the delivery of sediments, nutrients and pesticides to the end of catchments. These models are supported by **sub-catchment and catchment monitoring** information for model validation at key locations in sub

catchments and at end-of-catchment locations. The models included in this program can take into account trapping on floodplains, system lag times and biological processes (i.e. denitrification) and include SedNet (e.g. McKergow *et al.*, 2005) and WaterCAST (Water and Contaminant Analysis and Simulation Tool – formerly E2 and currently being refined into 'Source Catchments'[5]; Cook *et al.*, 2009). While SedNet can be used in this situation at a catchment scale and has routines which account for sediment trapping and other in-catchment processes, it is a long-term, time-averaged model which does not explicitly model system dynamics such as vegetation change, and so has limited usefulness for predicting changes in pollutant loads annually. However SedNet is powerful at identifying the spatial sources of suspended sediment and nutrients within the catchment, and hence can be compared to monitoring data at a number of scales, e.g. at small sub-catchment scales (e.g. Bartley *et al.*, 2007), at catchment scales (e.g. Wilkinson *et al.*, 2009) and in large river basins (e.g. Bainbridge *et al.*, 2007; Fentie *et al.*, 2005).

WaterCAST can produce annual loads due to its short time-step capabilities and is being developed to represent catchment trapping mechanisms and dissolved nutrients. Pesticides are currently not included in SedNet or WaterCAST approaches, and a simple model using catchment management support system (CMSS) concepts (e.g. Davis *et al.*, 1998) is included, where export coefficients for individual land use under specific management practices are aggregated to the catchment scale.

The role of over-bank floods in delivering sediments and nutrients to the GBR lagoon has now been quantified by MTSRF-funded researchers (Wallace *et al.*, 2009a, 2010a, 2010b) using a new **Floodplain Hydrodynamic model**. This model has only been applied in the Tully-Murray catchments and further applications in contrasting catchments (e.g. in the Dry Tropics) are needed.

Information on **catchment condition** including groundcover and wetland extent and condition is also reported, and information to support management of **catchment waterway and instream health** could be incorporated at this point.

Linking end-of-catchment loads with marine trigger values requires a receiving water model to simulate the fate and impacts of these contaminants as they pass through estuaries and into the GBR lagoon and beyond. MTSRF-funded research has contributed to the development of a **GBR-wide hydrodynamic model** to support achievement of the overall program objective of ultimately assessing or predicting the response of the GBR to improved land management, in conjunction with the supporting **Marine Monitoring Program**. **Estuary monitoring** is currently limited to two catchments in the GBR – the Fitzroy and the Burnett – where estuary processing is important for material delivery.

Integration and alignment of the components of the program are critical to permit reporting against the Reef Plan and Reef Rescue targets. However, linking the entire series of monitoring and modelling steps together through a GBR-wide assessment approach can be difficult due to the propagation of error and uncertainty between the individual steps (Brodie *et al.*, 2009b). One way to address some of these issues is to use a Bayesian Belief Network as a model integration tool. These approaches are currently being developed within the GBR region and at the GBR-wide scale (e.g. Lynam *et al.*, 2010b; Shenton *et al.*, 2010), and could be considered in the development of the reporting framework for the Reef Plan and Reef Rescue.

The development of reliable and effective indicators for monitoring and evaluation of the ecosystems of the GBR and its catchments depends on comprehensive understanding of

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[5] http://www.toolkit.net.au/watercast/

ecosystem function, and relationships within each ecosystem at multiple scales, between the system components of the catchment and marine systems. The following section provides a brief overview of the current knowledge about the impacts of various water quality parameters on GBR catchment and marine ecosystems that has been established through the MTSRF, and identifies how this information has been transformed into indicators of water quality and ecosystem health to inform monitoring and evaluation programs for the GBR and for tropical reef systems more broadly.

4.2 Management practices (plot/paddock scale)

Monitoring and modelling at the plot/paddock scale have generally received less attention through the implementation of the Reef Plan, however, as investment in on-ground action has increased, the need for information about adoption rates and effectiveness of specific management practices is widely recognised. A preliminary overview of the current knowledge of the impact of management on plot/paddock scale water quality was completed by Conics in 2009 (Freebairn, 2009), providing the basis for justification for further monitoring in specific industries and locations, and documentation of input data to paddock-scale models such as APSIM and HowLeaky. The information also supported the definition of a framework for management practices in the GBR catchments (see discussion below), and identified a set of pollutant generation rates for different management practices, by industry and by region.

The focus of MTSRF research at this scale has been on the identification of best management practices in Wet Tropics and Dry Tropics case studies (in terms of water quality benefit and cost effectiveness), and using these as input data to integrative models for assessing the water quality benefits and economic outcomes of management options under various scenarios (e.g. van Grieken *et al.*, 2010a; 2010b). The findings have fed into the design of the Paddock to Reef Program, and provide useful tools for testing different management scenarios for target setting.

4.2.1 System understanding

Information related to system understanding at this scale is locally and regionally specific and, therefore, is considered to be outside the scope of this report. However, relevant information is included in the Wet and Dry Tropics case studies in the companion report, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.).

4.2.2 Indicators for monitoring and evaluation

Paddock-scale water quality runoff

Over the last five years a range of research has been conducted at the paddock scale to investigate the impact of different land use practices and management on the water quality of receiving waters, however, little of this has been funded through the MTSRF, with the exception of small components of several projects led by the Australian Centre for Tropical Freshwater Research (ACTFR) (including Brodie, Bainbridge, Davis, Faithful and Lewis). In general, paddock-scale water quality runoff studies have tended to focus on one particular land use, and have often been established through regional collaborative arrangements between interested growers/graziers, key industry groups (BSES, Canegrowers, GrowCom), Federal/State Government initiatives and associated extension staff (such as SRDC, AgSIP and DPI&F (now DEEDI) Reef Extension Project) and research organisations (CSIRO and universities). The WQIP process built on and contributed to existing paddock-scale research projects, and provided broader frameworks that have and will continue to aid in the rollout of

the Best Management Practices (BMP) and further water quality plot-scale research to support the Paddock to Reef Program.

Most paddock-scale research has focused on the more intensive land uses such as sugarcane and horticulture, for example, Douglas Shire trial of N-fertiliser use in sugarcane (Webster and Brodie, 2008); Tully plot-scale runoff for cane and horticulture (banana) industries (Faithful *et al.*, 2006); pine plantation runoff (Faithful *et al.*, 2005); Lower Burdekin cane BMPs (Thorburn *et al.*, 2009; Thorburn and Attard, 2007; Thorburn *et al.*, 2007a, 2007b; Hesp, 2006); BSES Lower Burdekin sugarcane paddock runoff (Ham, 2006); Burdekin WQIP pesticide study (Lewis *et al.*, 2007b); Mackay-Whitsunday cane BMP trial (Masters *et al.*, 2008); and AgSIP Bundaberg (Stork *et al.*, 2007). Monitoring techniques have improved in recent years to capture event flows at this scale through more specific and targeted monitoring design, consistent application of sample collection and analysis techniques, greater landholder engagement, and an improved understanding of the scale of response relevant to paddock-scale water quality (e.g. Lewis *et al.*, 2007a; Bainbridge *et al.*, 2006a, 2006b, 2008; Armour *et al.*, 2006; Faithful *et al.*, 2006; Ham, 2006).

Paddock and small catchment-scale research has also been conducted on the water quality impacts of the grazing industry. Long-term research projects, e.g. DPI&F Wambiana grazing trials (O'Reagain *et al.*, 2005) and Virginia Park (Bartley *et al.*, 2007) in the Burdekin catchment are building on previous studies that investigated the relationship between ground cover and runoff water quality in semi-arid rangeland systems. Improved monitoring techniques similar to those identified above have also been established through these projects (e.g. Bainbridge *et al.*, 2009b) and are being applied in the Paddock to Reef Program.

Cost effectiveness of management practices

Socio economic research into the cost effectiveness of the recommended management practices for the major industries (sugarcane, horticulture, grazing and forestry) within the GBR catchments has been conducted across the entire region (van Grieken *et al.*, 2008, 2009, 2010a, 2010b), and in more detail for the Douglas Shire and Tully-Murray WQIP processes (Roebeling and van Grieken, 2009; Roebeling *et al.*, 2007a, 2007b, 2009a, 2009b; Roebeling, 2006; Bohnet *et al.*, 2006, 2007). This approach also incorporates production system simulation models (such as APSIM) and the SedNet and ANNEX models (Roebeling *et al.*, 2007a; 2007b; 2009a; 2009b; van Grieken *et al.*, 2010b; Roebeling and van Grieken, 2009), thereby allowing for the evaluation of management practice adoption (described in more detail below).

Management practice adoption

A suitable indicator for assessing the performance of management interventions is the adoption rates of various practices and tracking extension efforts by region and industry. At this stage, limited effort has been made to benchmark these indicators (with some exceptions at a regional scale). This is critical information needed to inform the evaluation of Reef Plan and Reef Rescue investments. In addition, in order to influence change in land management practices, managers need to understand landholder behaviours, their capacity to change and the likelihood of changes being implemented. There are multiple economic and social impediments to the implementation of changes of management practices aimed at reducing contaminant loads to the GBR. While 'win-win' scenarios exist for some management interventions (such as the '6 Easy Steps' nutrient management system in sugarcane) many practices involve net costs to producers, particularly in the shorter term. Economic and social impediments to practice change vary between regions, complicating the design of policies to achieve practice change (Brodie *et al.*, 2009b). These impediments have been studied in various locations, including the Tully-Murray catchment. Based on the farmer profiling done by Bohnet (in van Grieken *et al.*, 2009) the following results represent farmers'

likelihoods to adapt to management practices or management practice systems in the Tully-Murray catchment. Table 3 classifies farmers and their suggested ability and/or willingness to change. Change is defined as converting to a management practice that is improving water quality. This information can be used as an indicator in the prediction of the likelihood of change, or adoption of improved practices, relevant to target setting. Profiling of this nature across the GBR catchments would enable managers to identify areas of investment priority and provide insight into the type of management strategies that may be most successful in a particular location or farming community. This is also a priority research area for the Reef Rescue Research and Development Program.

Table 3. Farmers' profiles (typology) and ability and/or willingness to change in the Tully-Murray catchment (from van Grieken *et al.*, 2009).

Profile	Farm type	Change	Description
Large sugarcane farmers	1	++	The large sugarcane farmers are willing to adapt to improved management practices in the case where it will provide them with financial benefits (win-win) as well. They have a knowledge seeking attitude. From an economic point of view, their behaviour could be described as profit maximising.
			In response to policy interventions, farmers with this profile would have a positive attitude towards incentive schemes such as pollution control subsidies, even if this would mean 'sharing the burden'.
			Large sugarcane farmers look closely at how the business operates and therefore know what their costs are; often on a per tonne basis (e.g. they know their fertiliser costs per tonne and what proportion it makes up of the total cost per tonne). Large sugarcane farmers are business operators driven by income maximisation via cost minimisation.
Medium sugarcane farmers	2	+	The medium sugarcane farmers have a more sceptical view of management practices. Of all sugarcane farmer profiles, those with a medium-sized operation seem to experience the greatest trade-off between labour and leisure time, where sacrificing leisure time to labour for changing management practices can be seen as a constraint.
			These farmers are traditionalists, being a sugarcane farmer drives them. Their biggest source of information is other cane farmers.
Small sugarcane farmers	3	-	This profile represents farmers with multiple restrictions, such as on capital investments and labour (these farmers work off-farm, so in order to invest time into changing practice; they need to hire labour, which comes at a substantial cost). In the past, this profile was likely to sell their land to forestry companies or other sugarcane farmers if restrictions were too binding.
			Small sugarcane farmers' successors may not take over the business from their parents. Therefore, it is possible that some of these farms will be sold in the future.
			Because the small sugarcane farmers work mainly off-farm, exposure to extension material or time to learn about better farming techniques is limited.

Profile	Farm type	Change	Description
Mixed crop farmers	4	++/+	The farmers in this profile use diversification to increase flexibility towards market fluctuations and other risks. Often different members of the family supervise a different cropping system and, since every family member works on-farm, labour is hardly seen as a constraint. It must be noted that leisure time is valued so some restriction to labour intensive management practices must be incorporated.
			The mixed farmers are considered to be willing to change in case of win-win situations and also because improved water quality could lead to 'green labelling' and improve public opinion towards their business.
			Growing bananas requires high-level precision farming, which translates to sugarcane farming. These farmers will actively look for information but are able to critically evaluate it and decide if new ideas will work. Being a good farmer is what drives them.
Graziers	5	-	The graziers are seen as very independent traditionalists. Their biggest source of information is other farmers. Being a grazier drives them.

Management practice targets

As identified above, indicators related to management practices at the paddock scale relate to adoption rates, water quality and economic outcomes. Assessments that incorporate cost-benefit and water quality outcomes of particular management practices can assist in prioritising the adoption of practices and predicting outcomes of a set of management actions. There has been considerable MTSRF-funded effort in this research area in collaboration with CSIRO (e.g. Roebeling, van Grieken, Bohnet), with a focus in the Wet Tropics and Burdekin regions.

Evaluation of the performance of management practices across industries and regions first requires a set of agreed standards, which can then be used to set targets. A management practice framework developed through the MTSRF (van Grieken *et al.*, 2010a) was adopted from the 'ABCD' framework of management practices classification in use by NRM regions for the Reef Rescue and Paddock to Reef programs. The ABCD framework was originally adapted by Reef Catchments NRM (formerly Mackay-Whitsunday NRM) to sugarcane from the Queensland Department of Employment, Economic Development and Innovation (DEEDI) grazing land assessment tool. Since publication of the framework, other NRM regions have joined together with reef catchments to adopt the framework and standardise management practices into a statewide framework adopted by all of the regions, and it has been extended to cover land uses other than sugarcane, drawing on information generated through the research of Roebeling and van Grieken (and teams) under the MTSRF. The framework is a planning tool and is determined by criteria relating to:

- The resource condition achieved by adopting the management practice in the short, medium and long term;
- The acceptability of the management practices to the community; and
- The feasibility of achieving widespread adoption of the management practices in the short, medium and long term.

The ABCD framework provides a standard definition of management practices and a four-step scale (D-C-B-A) of improvement from 'old' to 'cutting edge' management practices. In the framework, the class A, B, C or D represents a *suite* of individual management practices.

Having a suite of practices for each class allows the modelling framework to define each one and run simulations for these management practices. Further detail of the framework is described in van Grieken *et al.* (2010a) and discussed in the companion report, 'Priority Pollutants in the GBR' (Waterhouse and Brodie, in prep.). Using these classes, targets for adoption can be set, for example, achievement of a proportion of landholders (x%) achieving x class practices by 2013.

To evaluate these targets, environmental economic models have been developed to explore which policy instruments are likely to be most effective in stimulating adoption of prioritised land management practice changes (Roebeling *et al.*, 2007a; van Grieken *et al.*, 2008, 2009). Adoption rates are assessed by predicting the likelihood that different types of farmers will respond to different policy interventions. Furthermore, the corresponding regional socio-economic and environmental consequences of implementing the policy are estimated, such as changes in local income, employment and nitrogen runoff.

The approach is based on the integration of a financial-economic analysis of actual and best agricultural management practices in sugarcane, beef cattle production, banana cultivation and production forestry at the plot level (see Roebeling *et al.*, 2009a), and a private economic farm household modelling approach for key agricultural producers at the farm level (see for example Roebeling *et al.*, 2000). This private-economic analysis not only provides insight into the likelihood of adoption of best management practices by agricultural producers and subsequent private-economic and water quality effects, but also enables the identification of incentives and regulations that are likely to be most (cost-) effective in promoting the adoption of best management practices.

Policy scenarios include regulations on water quality improvement (via DIN delivery) and fertiliser input, incentives for reducing fertiliser input (subsidies for reducing fertiliser input) and a taxation instrument to reduce fertiliser use. Furthermore, the situation can be simulated where all farm types would manage their crops via the preferred code of practice (it must be noted that the costs of enforcing and monitoring this situation are not included in the analyses).

Further detail of these studies is provided in the companion report on water quality and climate change interactions and socio-economic influences on management (Martin, in prep.).

Improved production system simulation and catchment hydrological modelling can assist in better estimations of the amount of DIN available for delivery to the waterways. The data used in the analysis need to be re-evaluated and more attention should be paid to uncertainty in the modelling components and the provision of risk associated with specific practices. In addition, transaction costs (hence, barriers) need to be investigated and quantified more deeply, such as the monitoring of the exact costs of change (training, machinery and other capital requirements, land conversion, etc.). Extended socio-economic surveying could provide more information on the exact risk aversion of farmers, which could be a restriction towards change.

4.2.3 Conclusions

Recording the adoption of defined classes or types of management practices is critical for measuring progress of land management improvement in the GBR catchments. Models can then be used to predict the water quality and economic outcomes of these improvements at various scales.

Financial motives are important to explain adoption of management practices. However, there are other non-financial factors explaining the non-adoption of management practices by farmers. The presented research addresses this by incorporating heterogeneity among landholders and allows analysis of a range of policy scenarios to test the cost-effectiveness of management practice change and improving water quality in the region.

The following indicators are recommended for monitoring and evaluation of management practices for water quality management in the GBR:

1. Types/ classes of management practices and rates of adoption;
2. Cost effectiveness of management practices; and
3. Capacity to adopt new or improved practices through landholder profiling.

In addition, assessments that incorporate cost-benefit and water quality outcomes of particular management practices can assist in prioritising the adoption of practices and predicting outcomes of a set of management actions.

Furthermore, aggregation of farm-level results to the catchment level has increased understanding and insight of potential regional income and water quality effects (e.g. Roebeling *et al.*, 2007b, 2009a). Inclusion of regional constraints which account for, for example, lower bounds on sugarcane supply to the Tully mill, provide economic indicators for social change. The inclusion of downstream costs for the tourism and fishing industries resulting from terrestrial water pollution and reef degradation (Wooldridge, 2009) allows a better representation and understanding of the linked socio-economic system.

4.3 Catchment and instream health

4.3.1 System understanding and thresholds of concern

Aquatic ecosystem health has long been regarded as a function of water quality alone – this is certainly more the case for marine waters. Within catchments, water quality is only one of the issues affecting ecosystem health, and not necessarily the most important. Habitat integrity, riparian condition, flooding and levels of weed infestation can all have a major bearing, as is the case for Wet Tropics streams (see Pearson, 2005; Arthington and Pearson, 2007; Mackay *et al.*, 2010), and as may be the case for coastal wetlands in the GBR catchments, in particular the Tully-Murray wetlands (Pearson *et al.*, 2010a, 2010b; Wallace *et al.*, 2010b; Karim *et al.*, submitted, 2010a, 2010b). Our understanding of the ecosystem health of GBR waterways has been greatly enhanced by recent reports generated through the Catchment to Reef Joint Research Program and the MTSRF on Wet Tropics streams (e.g. Arthington and Pearson, 2007; Pearson and Stork, 2007; Connolly *et al.*, 2007a, 2007b; Mackay *et al.*, 2010; Pusey *et al.*, 2007a, 2007b; Faithful *et al.*, 2006) and floodplain waterways (Pearson, 2005; Pearson *et al.*, 2010a, 2010b; Wallace *et al.*, 2009a, 2010a), and on the riverine waterholes and floodplains of the Dry Tropics (e.g. Perna and Burrows, 2005; Blanchette, 2010).

Water quality and water quantity (e.g. high and low flows) can have a major effect on aquatic ecology. From a water quality perspective, ambient or chronic water quality is of greatest importance to the ecology of the rivers and wetlands, as opposed to the short-term events that appear to drive water quality in coastal waters (Arthington and Pearson, 2007), and different monitoring strategies are required to address these different issues. Most of the contaminants associated with major storm events pass through the freshwater system so rapidly that they have little effect on stream ecosystems. Conversely, concentrations of materials measured in ambient conditions indicate the conditions that persist for much of the year and that have greatest influence on the health of stream ecosystems. During this period, major interactions between microbes, plants, animals and water chemistry occur, and water chemistry and biology are closely connected (Brodie and Mitchell, 2005, 2006). During the wet season flood pulses have important effects on river and wetland ecology (Junk *et al.*, 1989; Poff *et al.*, 2009). The first flood pulse can have high concentrations of organic carbon, suspended particulate matter, nutrients and hence biological oxygen demand, and this can cause hypoxia and fish kills (Pearson *et al.*, 2003). Thus, from a water quality perspective, flood events are typically most important with regard to contaminant exports, while the intervening periods are more important to the ecology of waterways. Moreover, perennial streams with constant dilution of local effects and with a strong influence of whole-catchment effects contrast substantially with waterholes and wetlands in which flow is intermittent and local effects predominate. Floods and flow regimes in general also have a major influence on aquatic health. For example, over-bank flooding has been shown to control wetland connectivity (Karim *et al.*, submitted, 2010a, 2009a, 2009b, 2008) which has a major influence on fish populations. This hydro-ecological connection has been demonstrated by Godfrey (2009) and Godfrey *et al.* (2010), who have shown that the relationship between the structure and dynamics of the larval fish assemblage in lowland riverine Wet Tropics habitats and the underlying variability of the habitat and its condition are shaped primarily by the prevailing flood regime. The most serious factors currently affecting health in Wet Tropics streams and wetlands are changes to habitats, including flow modification (e.g. due to the introduction of drainage networks), invasion by exotic weeds and loss of riparian vegetation, which can cause major changes to waterway morphology, habitat complexity, food availability, gas exchange with the atmosphere and, therefore, biodiversity (Arthington and Pearson, 2007). Organic effluents have been shown to cause fish kills and a major decrease in biodiversity as a result of oxygen depletion, while deposition of fine sediments derived from agriculture and other sources reduce biodiversity in streams (*ibid*). Dry Tropics streams

and wetlands are affected by similar influences but due to varying land uses and a dominance of cattle grazing, are generally more exposed to issues related to sedimentation. Many Dry Tropics rivers cease to flow in the dry season, contracting to isolated lagoons, which provide refugia for the biota (Pusey *et al.*, 1998). These lagoons develop their own character depending on their lithology, riparian vegetation, cattle access, etc. Their nature is very different from the lagoons of the Wet Tropics, and the determinants of ecosystem health within them are likewise quite different. Meanwhile, those water courses that continue to flow (e.g. the main Burdekin River, much of the time) are shallow, warm and highly productive – probably much more so than Wet Tropics rivers. But boom can be followed by bust as even the most reliable Dry Tropics rivers can cease to flow in very dry years (Pusey *et al.*, 1998). The gradient of flow regime from mid Wet Tropics to Dry Tropics is very clearly reflected in their biodiversity, with even the smaller rivers of the Wet Tropics supporting many more fish species than the large Dry Tropics systems of the GBR catchment (Pusey *et al.*, 2007a).

Disturbance of riverbanks is also caused by feral animals, including several species of fish, such as tilapia (Webb, 2006), and pigs, which can severely disturb the sediments and benthic fauna of shallow wetlands. In the Wet Tropics, of major concern are tilapia (*Oreochromis mossambicus*) and other related cichlid fishes of African origin that, it is feared, might displace native species (Burrows, 2004; Canonico *et al.*, 2005). Currently it appears that introduced fishes do especially well in disturbed habitats, but are not yet implicated in displacement of native species in less disturbed systems (Bunn and Arthington, 2002; Webb, 2003).

The loss of riparian (riverbank) vegetation in the GBR catchments is documented throughout Queensland in the Statewide Landcover and Trees Study (SLATS) (e.g. Queensland Department of Natural Resources and Water, 2007), and at a local or regional scale through specific, mostly short-term, assessments. Natural riparian vegetation in GBR catchments typically includes forest trees, shrubs and, with sufficient light penetration, some grasses and herbs. Where drainage is poor, species that are tolerant of waterlogging may dominate. The benefits of riparian vegetation to normal ecosystem function are well documented (e.g. Pusey and Arthington, 2003). They include: habitat and habitat corridors for terrestrial animals and plants; habitat for semi-aquatic animals; shade; filtration mechanisms; organic inputs; bank stability; instream habitat via roots and snags; basking sites for reptiles; and breeding and roosting sites for many partly aquatic species, ranging from insects to birds.

The dynamics of oxygen (and, incidentally, pH) in catchment waterways are complex and depend on a range of natural and human-influenced variables (Pearson *et al.*, 2003). Natural oxygen status can best be achieved by maintaining normal flow regimes and riparian zones; by curtailing weed growth; by preventing the input of nutrients; and by removing blockages to flow. While the tropical Australian invertebrate and fish fauna appear extremely resilient to low dissolved oxygen status (Pearson *et al.*, 2003; Connolly *et al.*, 2004), their tolerance thresholds can be breached, as evidenced by the occasional fish kills that occur in floodplain waterways. Prolonged high sediment levels reduce diversity and abundance of stream biota such as fishes (Hortle and Pearson, 1990).

Organic inputs to aquatic systems, such as effluents from sewage works or dairies, typically cause oxygen depletion through bacterial respiration of organic materials, with subsequent loss of hypoxia-intolerant species of invertebrates and fish. In the Wet Tropics, sugar mill effluents were once the main source of problems (Pearson and Penridge, 1987), but there has been substantial effort to remove or clean up discharges to waterways. It is likely, however, that organic inputs from harvested cane fields still can create adverse conditions in waterways, especially when flows are insufficient to flush poor quality water out of the system (Arthington *et al.*, 1997).

Thresholds of concern

As supporting information to the development of thresholds of concern for freshwater ecosystems in the GBR, MTSRF research aimed to measure spatial and temporal variability of biophysical indicators in floodplain lagoons along natural environmental gradients and gradients of disturbance (see Pearson *et al.*, 2010a for an overview of this research). In particular, the field study in the Tully-Murray catchment was designed such that stressor-response relationships along gradients of disturbance (supported by data from laboratory trials and the literature) would help to identify thresholds – points along each disturbance gradient where ecological changes of scientific or management concern become apparent.

Previous work in the Wet Tropics has documented some thresholds for selected species and variables, including dissolved oxygen (Pearson and Penridge, 1987; Connolly *et al.*, 2004 – see Figure 2), nutrients (Pearson and Connolly, 2000), ammonia (Økelsrud and Pearson, 2007), substrate disturbance (Rosser and Pearson, 1995), and sediment deposition (Connolly and Pearson, 2007). Recently a large project by the Australian Centre for Tropical Freshwater Research (ACTFR) documented critical levels of dissolved oxygen and guidelines for many species of tropical Australian freshwater fish (Butler *et al.*, 2007; Butler and Burrows, 2007). This type of work is essential for understanding species' responses and thresholds, and in the case of the ACTFR work, developing guidelines against selected criteria (in this case, dissolved oxygen).

Figure 2. Threshold pattern for dissolved oxygen and several taxa from Wet Tropics streams (from Connolly *et al.*, 2004).
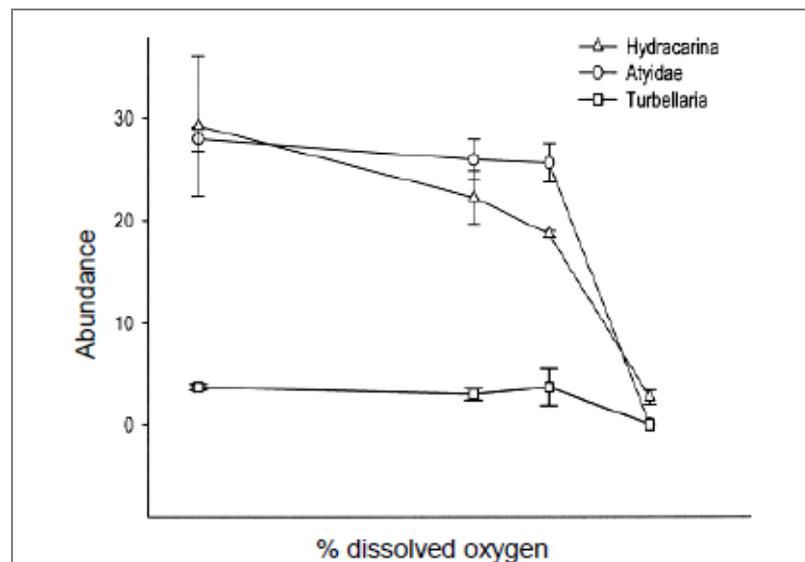
However, in ecosystems there are multiple factors affecting species, some of which act along similar axes, but others of which act independently. Some factors may have no effect while others have a linear or gradual effect such that no clear threshold exists (Figure 3). In the wetland situation, for example, it is possible that light levels have a direct linear effect on plant abundance, but plants will also be affected by changes in habitat, nutrients, concentration of herbicide, etc., which may have non-linear effects (Mackay *et al.*, 2010). In the Tully-Murray many habitat and water quality variables were measured, each of which may have independent effects on each species of plant and animal. Moreover, many variables will act differently on different life stages of the biota, so the end result is a composite response to these multiple effects. In some situations a single variable overrides all others. Dissolved oxygen is one such variable that can control presence or absence of fish (as in Figure 2).

Figure 3. Some alternative threshold patterns for different indicator and pressure variables (from Pearson *et al.*, 2010a).
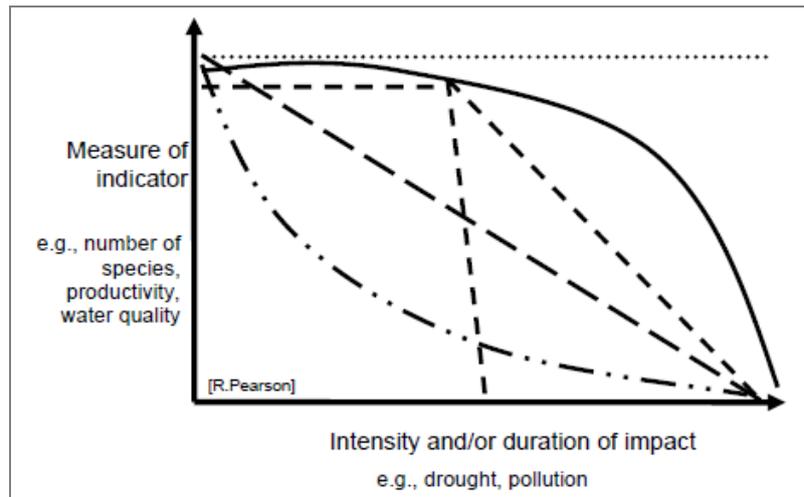
The Tully-Murray analyses indicate, however, that many water quality variables and habitat variables act in concert, such that there were significant relationships between ordination axes and many of the variables. It is thus clear that the multiple responses to multiple variables are expressed quite generally. It is evident that in the Tully-Murray Wetlands, as in the Wet Tropics stream study (Arthington and Pearson, 2007) many of the variables that affect the biota can be resolved into 'habitat' and 'water quality' composites. Thus, a gradient of response to a gradient of land-use impact is evident, defined by the variable composites. The fact that response to the various factors measured can be identified indicates that some composite threshold has been crossed. Without reference sites it is difficult to establish where that threshold might be situated (Figure 4). The best guess at sites closest to reference condition are notional lagoons 1 and 6; but in this schema they are somewhat removed from reference (unknown how far). This is how the Tully-Murray wetlands are perceived. The goal in management is to have sites progressively take a trajectory up and beyond the notional 'threshold of concern'.

Figure 4. Notional position of Tully-Murray sites on an impact/location gradient (from Pearson *et al.*, 2010a).
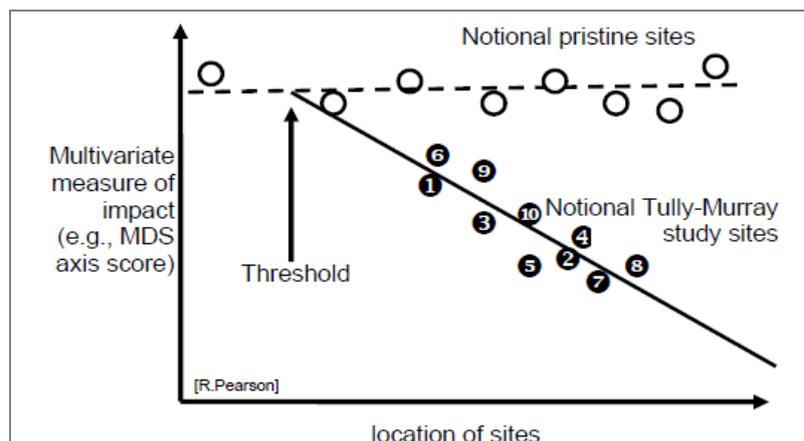
Clearly, measuring such a threshold in the wetlands is not as easy as in the oxygen experiment (Figure 2). Furthermore, some parts of the gradient are entirely natural (e.g. distance from the coast; salinity) although even they may be affected by development (distance to the coast along channels might be greatly reduced by drainage works). Therefore, natural gradients need to be removed from consideration, as done in the stream study (Connolly *et al.*, 2007a; Pusey *et al.*, 2007b). In that study multiple replicate sites, and comparisons between more and less impacted catchments, facilitated analysis and

interpretation. In the Tully-Murray wetlands the study was very constrained by the low number of sites available, creating a gradient of sites that was truncated at both pristine and disturbed ends of the spectrum, so the interpretation is necessarily more equivocal. Nevertheless, the results provide a robust set of stressor-response relationships and a substantial benchmark against which improvement in the ecological condition of floodplain lagoons can be evaluated.

4.3.2 Indicators for monitoring and evaluation

Instream health

Water quality variables themselves do not necessarily relate directly to the system's health (e.g. normal biodiversity and ecological processes). For example, enhanced nutrient levels do not directly affect invertebrates or fish – it is only through interlinked processes that effects are felt (Pearson and Connolly, 2000; Pearson *et al.*, 2003; Kennard *et al.*, 2006a, 2006b). Therefore, contemporary assessments of river health incorporate both physico-chemical measures and measures of ecological integrity. Habitat integrity, riparian condition and levels of weed infestation can all have a major bearing on aquatic ecosystem health, as shown in previous research on Wet Tropics streams (Werren and Arthington, 2002; Arthington and Pearson, 2007; Connolly *et al.*, 2007a, 2007b; Mackay *et al.*, 2010), and this may be the case for coastal wetlands in the GBR catchment, such as the Tully-Murray wetlands.

Different indicators can be expected to reveal different aspects of stream health but a comprehensive monitoring protocol requires the incorporation of a suite of physical, chemical and biological measures into an integrated framework that could be used to assess the health of stream systems relative to reference conditions, measured pressure and known disturbances. Research detailing the chain of influence from land use to stream ecosystem response, via the responses of individual ecosystem components, is required to understand how these influences operate and to underpin the development of monitoring tools and guidelines appropriate to Wet Tropics streams and wetlands.

Research undertaken as part of the Catchment to Reef Joint Research Program was designed to explore the concept of river health and to represent it as an integrated suite of protocols and techniques for biological river health assessment in Wet Tropics streams. The ultimate goal was the adoption of the methodology by relevant agencies and persons responsible for or interested in ecosystem health monitoring. The technical output of this research was summarised in Arthington and Pearson (2007).

Further detailed work by Pearson, Wallace and others into suitable indicators of freshwater ecosystem health in Wet Tropics streams and floodplain wetlands indicates that ecosystem health indicators clearly respond to hydrological, physico-chemical and habitat variables (Arthington *et al.*, 2005; Pearson *et al.*, 2010a; Wallace *et al.*, 2010b; Karim *et al.*, 2010a). The biotic components of the ecosystem appear to integrate impacts and indicate the capacity of the ecosystem to sustain normal ecological communities and processes. There are distinct gradients of biotic assemblages, which correlate strongly with environmental parameters, variously ranging from plant abundance, through nutrient concentrations to concentrations of herbicides. Research on the Russell-Mulgrave system clearly demonstrated that fish communities responded to ecosystem health at the sub-catchment scale while invertebrates were good indicators at scales from stream reach to sub-catchment. For floodplain wetlands, analyses of a full twelve-month data set from the Tully-Murray wetlands will help determine whether cause and effect relationships can be linked with specific drivers; preliminary analyses suggest that several surrogates for land use will serve as indicators of both impact and ecosystem health. For example, it appears that fish are good indicators of ecosystem health at the lagoon scale, while invertebrates respond

more to fine-scale habitat features. Several taxa have very strong relationships with the dominant environmental axes and might well be developed as indicators, although where relationships are clear, the proximate cause of impact is often an easier indicator to monitor. One of the major tasks in indicator development is separating natural gradients, such as the change in species composition due to distance from the estuarine environment, from human-induced change, such as loss of species due to contamination by weeds or pesticides. Hydrological factors are of prime importance on the floodplain as they determine not only the permanency of waterways, but also their level of ecological connectivity across the floodplain – to other lagoons, rivers and the sea. This biological connectivity between the catchment and GBR lagoon is a factor that has not been adequately addressed in consideration of reef health, although several MTSRF projects will contribute important information as project results are finalized. For example, the key results from a hydrodynamic modelling study of wetland connectivity are summarised in the following section on wetland condition and connectivity.

Further detail of the progress of the selection of suitable indicators of catchment and instream health in the GBR catchments is provided below.

Water quality

Biological response to water quality parameters in the Wet Tropics case studies suggested strong influence of habitat (weeds etc), salinity, pH, dissolved oxygen, nutrients and pesticide concentrations (Pearson *et al.*, 2009). However, precise cause and effect relationships cannot yet be determined, as most parameters inter-correlate (positively or negatively). Generally, though, the correlations identify, on the one hand, natural gradients (distance from the coast, benthic substrates, etc.) and, on the other hand, human influences such as weeds, nutrients, pesticides and sediments. In many cases the connections are not straightforward: for example, invertebrates and fish are typically not directly affected by nutrients (e.g. Pearson and Connolly, 2000), but show strong correlations because of the cascading effects of nutrients on weeds, plankton, bacterial production, dissolved oxygen, etc. Standard water quality measures provide a snapshot description of environmental health, as well as suggesting sources of changes in biotic measure and are an important component of any ecosystem health assessment, as long as they are assessed following ecologically relevant protocols.

Nutrient concentrations in Russell-Mulgrave streams increased with the proportion of catchment devoted to agriculture (Connolly *et al.*, 2007a). This relationship is expected, and has been mirrored in the Tully catchment (Mitchell *et al.*, 2009). However, the relationship between nutrient concentrations, surface flow and groundwater dynamics is very complex and does not necessarily follow a simple input-output model. Nevertheless, it is clear that nutrient concentrations are greatly increased by agriculture to much higher levels than the Queensland guidelines, even in systems that might be regarded as having best-practice land management, and with largely intact riparian systems (Connolly *et al.*, 2007a).

Elevated concentrations of some pesticides have been found in Wet Tropics waterways, indicating that the concentrations of pesticides are highest in areas of intensive agricultural activity (e.g. Bainbridge *et al.*, 2006a) but there is very little information on the impacts of pesticides on native biota (exceptions include Kevan and Pearson, 1993). There are also circumstances where chemicals are applied directly to waterway weeds such as paragrass (Brodie *et al.*, 2008b). The implications of these applications for instream or downstream ecosystem health have not been adequately investigated. Some correlation between pesticide concentrations and biotic indicators has been shown in the Tully-Murray floodplain, but this result is confounded by other variables and any cause-effect relationship is not clear. Clearly, our understanding of the fate and impacts of pesticides on catchment and instream health is a major knowledge gap that needs addressing.

Macroinvertebrates

Whether or not their presence and abundance can be causally linked to particular environmental features, the presence of a 'normal' fauna is strong evidence of ecosystem health, and like the presence of any organism, evidence that conditions are appropriate to support that species or suite of species. Macroinvertebrates have long been regarded as the most efficient indicators of ecosystem health in freshwater systems. They represent very diverse taxa, with a diversity of responses to stressors; they are easy to sample; they tend to be sedentary so reflect local conditions; they integrate conditions over time (and are not a simple snapshot); they have relatively short life cycles so can show rapid population responses to changed conditions; and, with rapid assessment protocols and trained staff, they are relatively easy to process (Pearson *et al.*, 2009).

MTSRF-funded studies of Wet Tropics streams by Connolly *et al.* (2007a) showed clear patterns of macroinvertebrate distributions in the streams surveyed because of the strong gradient in substratum particle sizes along each stream and differences in particle sizes between streams. The macroinvertebrate assemblages were useful in classifying the streams into upper, middle and lower reaches and demonstrated a consistent longitudinal gradient of assemblage structure. The consistency of these patterns enabled comparisons between streams using analysis of covariance and this proved to be a robust approach in detecting differences between streams. Many samples were taken across gradients – an approach that proved to be very successful, and demonstrated to be of high utility in developing monitoring protocols. However, testing of indices and sample size demonstrated that to detect differences there was a trade-off in the amount of detail and effort applied at the site scale and the number of sites used in comparisons. Understanding this trade-off is valuable in that effort can be concentrated to suit individual constraints. Nevertheless, site selection will be critical to avoid confounding effects and will depend on prior knowledge of the macroinvertebrate distributions or require a pilot study.

Results describing the impacts of the loss of riparian vegetation and coarse particulate organic matter provide further evidence of the importance of riparian vegetation (Arthington and Pearson, 2007). The conclusions are tentative because the results have not been replicated in the catchments and similar surveys need to be carried out in different catchments to generalise the conclusions. Further surveys should encompass other types and degrees of impact to test the approach across different levels of disturbance. Nonetheless, the results showing that riparian vegetation is a key determinant in maintaining instream diversity are encouraging, as this is the most common remediation currently being applied in these streams and the results confirm that maintaining and rehabilitating riparian vegetation is a beneficial activity (Pusey and Arthington, 2003; Pusey *et al.*, 2007a; Mackay *et al.*, 2010).

The efficacy of different monitoring indices was demonstrated, with species richness being the most promising index, although identification at a higher taxonomic level (i.e. family) was still more effective than using the commonly adopted 'Stream Invertebrate Grade Number – Average Level' (SIGNAL) index. This highlights the need to test indices under the situations in which they are to be applied, and to ensure that appropriate measures are being used to answer the management questions.

In other studies, macroinvertebrates have been demonstrated to be sensitive to changes in water chemistry, including dissolved oxygen concentration (e.g. Connolly *et al.*, 2004), pH (e.g. Rutt *et al.*, 1990), salinity (e.g. Metzeling, 1993) and to be vulnerable to toxic contaminants such as insecticides (e.g. Liess, 1994; Shultz and Liess, 1995). They have also been shown to respond to organic pollution (e.g. Pearson and Penridge, 1987) and nutrient enrichment (e.g. Økelsrud and Pearson, 2007; Pearson and Connolly, 2000). The clearing of riparian vegetation and increases in sedimentation have also been shown to be detrimental

to macroinvertebrate assemblages (e.g. Ryan, 1991; Connolly and Pearson, 2007; Harrison *et al.*, 2008). Pearson and Penridge (1987) found high abundances but low biodiversity of macroinvertebrates below the outfall of a sugar mill in the Wet Tropics. They associated the increase in macroinvertebrate production with high levels of nutrients and organic matter in mill effluent. The pattern of abundance and diversity closely paralleled a well-known relationship between biota and organic effluents (such as sewage and animal wastes) in studies from the temperate northern hemisphere. In experiments using artificial stream channels on the bank of a rainforest stream Pearson and Connolly (2000) were able to increase macroinvertebrate abundance by 75% by increasing nutrient concentrations, but this treatment did not affect biotic diversity, indicating that impacts and responses can be quite subtle.

A MTSRF pilot study conducted in the Mackay-Whitsunday region (Leonard, 2009) investigated the efficacy of macroinvertebrates as a bioindicator in a predetermined land-use model developed by the regional NRM group (Reef Catchments) for the Mackay-Whitsunday Water Quality Improvement Plan. This study was designed as a pilot project to develop effective methods in which macroinvertebrate communities can be applied as bioindicators in a regional monitoring program. Macroinvertebrates were generally effective indicators of ecosystem health at the local scale but less so at the regional scale with this sample design. Local-scale variables such as riffles and pools, water velocity, and substrate size explained the greatest variance among sites and invertebrate communities. Previous studies agreed with these findings (Lammert and Allan, 1999; Johnson *et al.*, 2007). Catchment classes moderately, but still significantly, also explained some of the variance between invertebrate communities. The integrity of streams described by invertebrate and fish communities had mixed results, possibly because each taxonomic group responds to environmental variables at different spatial scales (Heino *et al.*, 2005). This supports the need for including multiple taxa in monitoring design (Lovell *et al.*, 2007; Rodrigues and Brooks, 2007), and difficulties of applying temperate (e.g. southern Australian) macroinvertebrate models in the tropics (e.g. northern Australia) (Chessman and Townsend, 2009).

Monitoring program methodologies that use macroinvertebrates as bioindicators in a land use model may be improved by accounting for local-scale variability. One improvement may be increasing the number of sites sampled along a lower number of creeks, as demonstrated by Connolly *et al.* (2007b). This method was useful in accounting for local environmental gradients, so clearer relationships could be discerned with land-use effects on ecosystem health (Connolly *et al.*, 2007b); without accounting for natural gradients, impacts were impossible to discern. In the Mackay-Whitsunday study, broad-scale sampling resulted in masking of the effects of land uses on invertebrate communities by local-scale influences. These influences played a greater role in defining invertebrate communities. Temporal differences, which were caused by flooding events during mid-February, were the only broad-scale variables that clearly influenced invertebrate communities between pre- and post-wet seasons.

Land use effects were primarily observed through correlations with two local-scale variables – riparian condition and degree of anthropogenic disturbance. These variables may have had the greatest association with catchments that had grazed and urban land uses. Sites at Basin Creek, in particular, did not fit the minimally impacted grazing catchment class definition (as defined in the WQIP) since they were distinct outliers that may require more immediate attention. Other grazed sites, such as Blacks Creek did not separate, due to the presence of riffles which contained high invertebrate richness and excellent riparian condition. Some discrepancies related to catchment classes and land uses in this study may have been caused by quantitative land uses and qualitative definitions in land use (Drewry *et al.*, 2006, 2008).

An improvement in monitoring design that is often touted is the use of reference condition rather than a reference site commonly used for comparisons of specific environmental gradients, such as land use (Wright *et al.*, 1984; Simpson and Norris, 2000; Connolly *et al.*, 2007a). Reference condition can be hard to achieve, but the paired catchment study, documented in a MTSRF report (Arthington and Pearson, 2007) demonstrates a very useful way forward. Reference sites may not be as useful in the Mackay-Whitsunday land-use framework since catchment classes were based on a continuous gradient of intensive cropping. If habitats were accounted for in a covariate analysis, an alternative solution may be to simply use a range of best possible conditions. A model could be developed where sites above a linear threshold could be considered in 'good' or at least 'better' condition (Figure 5). A goal would then be not to get sites above the line (which is impossible as it represents an average) but to raise this bar.

Figure 5. A possible model to be used with macroinvertebrate communities in the WQIP, showing a strong relationship with a habitat-level variable and a linear threshold. Conditions above the regression line are considered relatively good and below the line relatively poor. An increased number of sample sites along each stream may better show this linear relationship and reference condition. Source: Leonard (2009).
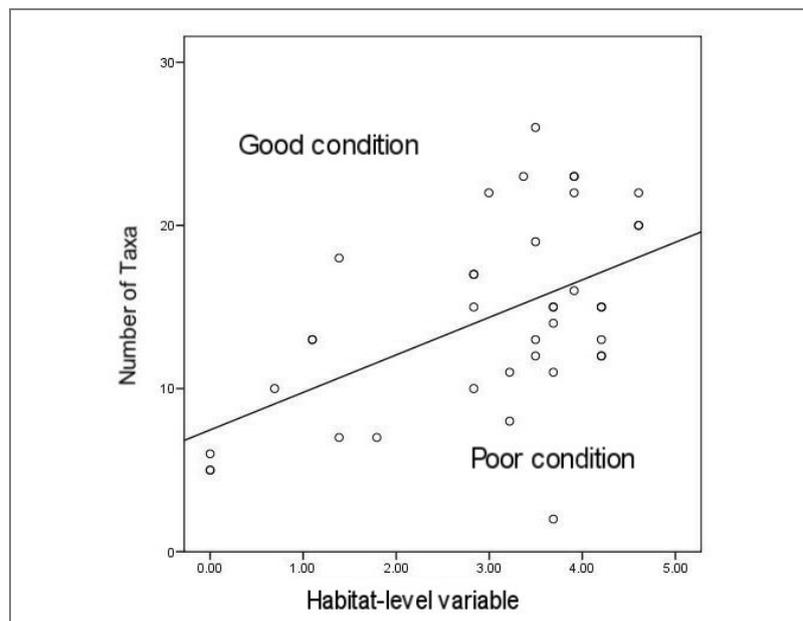
This approach is an adaptation of the Russell-Mulgrave study that used paired catchments (Connolly *et al.*, 2007a), in which one catchment represents reference (better) condition and the other represents poorer condition. Reference catchments in the Mackay-Whitsunday and other regions are not available and cannot be used in a broad-scale monitoring program, so the use of reference condition, with the benchmark of ecosystem health being the best conditions available, is the best pragmatic approach available.

In conclusion, the WQIP implemented by the Reef Catchments NRM Group can substantially benefit from the inclusion of macroinvertebrates as bioindicators of instream health. The study showed that environmental variables influence river and stream ecosystems within a hierarchal framework; therefore, it is important the monitoring design is developed within this framework. These findings have application in other locations and could be incorporated into a set of indicators for catchment and instream health in the GBR catchments.

Aquatic and riparian habitats

Macrophytes, whether they be submerged or at the surface, fixed or free-floating, provide habitat and refuge from predators for many freshwater species. Utilising them as ecological indicators has some merit when assessing ecosystem health, especially if the ratio of introduced to native species is assessed (Davis *et al.*, 1999), but overall their description and sampling is rather arbitrary when considering overall ecosystem health (Mackay *et al.*, 2003; 2010). Nevertheless, macrophytes were clearly a driver of macroinvertebrate community responses, and provide a useful and rapid method of assessment. Their presence or absence will also help explain differences in macroinvertebrate communities. A big unknown currently is how herbicides applied on land and in drains affect wetland plant assemblages. It is possible that different species are differentially affected by herbicides, with possible cascading effects on habitat structure and primary productivity (whether by phytoplankton, macrophytes or epiphytic algae), and so too zooplankton, macroinvertebrates, fish and, possibly, top predators such as birds and crocodiles.

Riparian condition is an important factor in aquatic systems, and it provided a strong signal in Wet Tropics streams (Connolly *et al.*, 2007a; Pusey *et al.*, 2007b). The macroinvertebrate communities were strongly influenced by the presence or absence of the riparian vegetation, largely, it would appear, through the amount of leaf litter in the stream substrate (terrestrial litter is well documented globally as a major food source for the invertebrate food web, and this is also true of Wet Tropics streams – Cheshire *et al.*, 2005). Absence of riparian vegetation not only leads to a loss of this major instream resource, but also typically leads to growth of invasive weeds such as para grass, which also show strong relationships with macroinvertebrates. Again, it is likely that multiple factors interact – while lack of shade or high nutrient levels may not affect macroinvertebrates or fish directly, they can have major effects on habitats and thereby influence the fauna.

The condition and extent of riparian areas is included as a Reef Plan target, specifically identifying improvements. While extent can be reported using vegetation mapping and satellite imagery, riparian condition is much more difficult to quantify. The above findings may assist in reporting against this target in the future. Riparian condition is one of the factors currently being considered in analyses of biotic communities in the Tully-Murray wetlands (see below).

Phytoplankton

Studies of instream phytoplankton populations in the Burdekin system (Preite, 2009) have demonstrated the close connection between phytoplankton community structure and dynamics, and the underlying variability of the habitat, whether due to natural or human influences. While the phytoplankton closely represents the status of ecosystem health, its utility in monitoring is limited by restricted knowledge of cause-effect relationships and because of the time-consuming nature of sample processing.

The results provide an improved understanding of phytoplankton dynamics in seasonal tropical rivers. They are patchy systems, driven by both predictable (seasonal) change and stochastic events (flood, drought). Water holes develop their own characteristic assemblages, according to local factors, but those within rivers are more similar than those between rivers, as a result of underlying differences in catchment characteristics. The influences of water quality and other variables therefore operate at catchment scales for part of the annual cycle, but at a local scale for the majority of it. The tolerance of phytoplankton species to environmental parameters, especially those that can be linked to river health, needs further investigation. Currently, the use of algal assemblages for regular or routine assessment of tropical river health is not practical because of their patchy and variable distribution.

Complementary work on invertebrate communities and food webs in the Burdekin system is giving a different perspective (Blanchette, 2010). Whether communities are random or highly structured in their assembly, especially after floods and during droughts, is very important to understanding their utility as indicators of health. The work has demonstrated that these indicators could be useful in these ecosystems. This project will provide the necessary conceptual and predictive model to allow development of robust monitoring protocols.

Fish

Fish are larger, longer lived and more mobile than macroinvertebrates so tend to be less useful as ecosystem health indicators at the site level (e.g. instream sites). However, they do respond to changing conditions across sites and are very good indicators at the catchment or sub-catchment level. Studies conducted by Pusey *et al.* (2007a) evaluated the extent to which present day agricultural practices and other anthropogenic stresses have an impact on stream fish in four sub-catchments of the Russell-Mulgrave River basin. The study involved a paired catchment comparison, with two streams sampled in both the Russell and Mulgrave catchments. The aim was to investigate the effects, if any, of contrasting land use and management practices in the two catchments, using both a comparative and referential approach to this question.

The relationships uncovered in this study, using analyses of factors affecting observed versus expected fish assemblage structure, all point to the value of using fish as indicators of stream degradation resulting from catchment land use and riparian degradation. Fish assemblages in Wet Tropics streams were particularly responsive to the effects of degraded riparian systems on stream habitat structure, especially aspects of habitat (e.g. velocity) related to the presence and abundance of aquatic macrophytes, including alien species such as para grass and Singapore daisy (Pusey *et al.*, 2007b). The presence and abundance of alien fish species were also correlated with altered habitat conditions, and were most prevalent and abundant in catchments with a high proportion of land use devoted to sugar cane production and urbanisation (Pusey *et al.*, 2007b).

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

Fish are also important as a major focus for monitoring in their own right (Kennard *et al.*, 2005, 2006a, 2006b). Several species that are commonly associated with the GBR lagoon and estuaries (including prized species such as barramundi and mangrove jack) spend substantial parts of their lives in wetland systems, so their presence or absence is noteworthy. They are dependent on good water quality, an appropriate habitat matrix and, crucially, on connectivity between different aquatic systems. Connectivity is affected by barriers due to restricted flow, infrastructure, weeds and poor water quality. Fish are therefore important as potential monitors of ecosystem health on a broad scale, well beyond the local site in a stream or floodplain lagoon. Patterns of distribution of several species in the Tully-Murray wetlands reflect these issues (Pearson *et al.*, 2010a).

Wetland condition and connectivity

Wetland systems in the GBR catchment have been greatly reduced in extent as a result of agricultural practice, roads and settlements (Pearson *et al.*, 2010a). The Reef Plan includes a target for wetlands in the GBR catchments that specifies no net loss or degradation of wetlands. This requires knowledge of wetland extent and condition. As part of the Queensland Wetland Program established in 2003 under the Reef Plan, Queensland's wetlands have been mapped digitally by building on existing information, including water body mapping derived from satellite imagery, regional ecosystem mapping and a springs database (EPA, 2005). Wetlands have been classified according to a range of criteria, including the type of ecological system (riverine, estuarine etc), their degree of water permanency, and salinity. The result is a consistent wetland map at a scale of 1:100,000, with finer detail in some parts of Queensland (mainly coastal regions) where appropriate mapping data exist. A wetland inventory is also being developed to describe the listing and storage of wetland information from a range of sources including tenure, climate, population, land use and field data. The MTSRF research team has made considerable contributions to the development of this inventory for wetlands in the GBR catchments.

Wetland condition also needs to be reported to demonstrate that there has not been further degradation of wetlands in the GBR catchments, and further work is required to fulfill this requirement of the Paddock to Reef Program.

Significant progress in defining how wetland ecological condition is affected by flood regimes has been made in MTSRF research conducted by Pearson, Wallace and others in the Tully-Murray catchments in the Wet Tropics (Pearson 2010a, 2010b; Wallace *et al.*, 2010b; Karim *et al.*, 2010a; Godfrey, 2009). The first quantitative estimates of wetland connectivity during and after flooding have been made using floodplain hydrodynamic models. These showed that during floods the duration of connection of individual wetlands varied (from one to twelve days) depending on flood magnitude and location in the floodplain, with some wetlands only connected during large floods (Karim *et al.*, 2010a). All of the wetlands studied were connected to the Tully River for shorter periods than they were to the Murray River, due to their proximity to the Murray River and the higher bank heights and levees on the Tully River (Figure 6). These variations in wetland connectivity could affect the movement of aquatic biota during flood events and the variability of habitat and biodiversity of individual wetlands. Flood pulses produce an initial connection between the wetlands, but after the inundation recedes is followed by a period of connectivity via the natural streams and the man made drains on the floodplain. Additional hydrodynamic modelling of the post-flood period (Figure 7; Karim *et al.*, 2010b) shows that wetlands located near the rivers and/or with good network connection maintain longer connection times with the rivers (Figure 7). Drainage network connectivity to both rivers varied from 30 to 365 days, and was much greater than flood inundation connectivity for the same wetlands (Figure 6). The connectivity of artificial wetlands varied greatly, from 10% to 100% of the year, according to the type of network connection they have; a result that has important implications for the location of these types of wetland. This MTSRF project has also shown how this kind of connectivity modelling can be used to identify when water levels in a drainage network fall below critical thresholds for fish movement using readily available river gauge data. These types of relationship are central to the concept of setting environmentally acceptable flows. Quantitative connectivity modelling will also be useful to help explain the variation in habitat structure, aquatic biota composition and water quality of individual wetlands over time.

These studies suggest that much of the fauna residing in the wetlands are resistant to the immense change in land use that has taken place across the floodplain in the last century. Potential indicators, such as plankton and benthic invertebrates, tend to correlate very closely with habitat changes, which themselves are quite straightforward to monitor (e.g. floods, riparian condition, exotic weeds). Fish appear to provide a more robust indication of overall wetland condition and, importantly, connectivity among wetlands and with rivers and tidal systems.

(a) Tully River
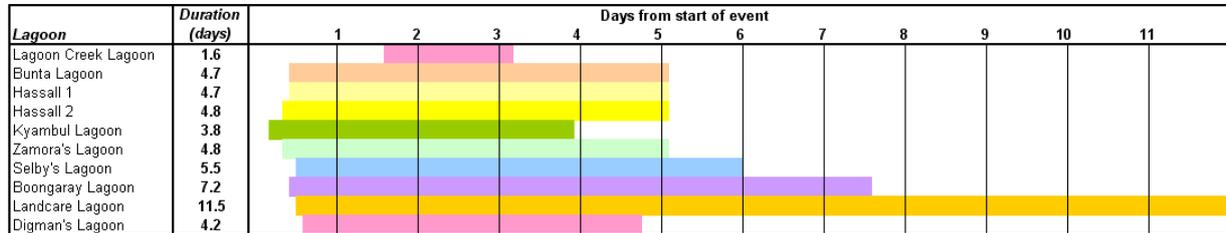
(b) Murray River
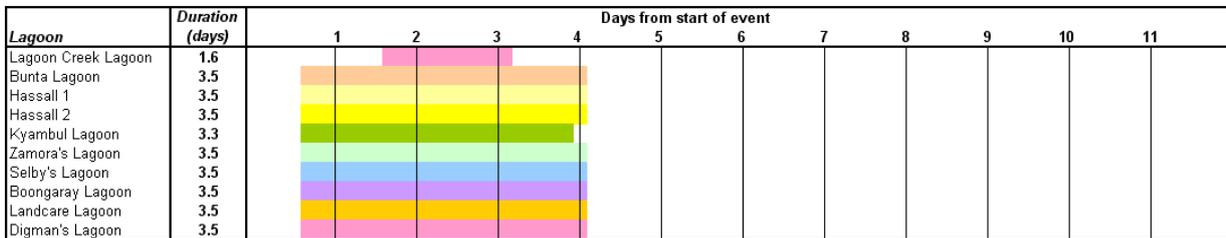
Figure 6. Summary of the timing and duration of connectivity of ten floodplain wetlands to the (a) Tully and (b) Murray rivers during flood events, with an annual return period of 20 years (from Karim *et al.*, 2010b).

(a) Tully River
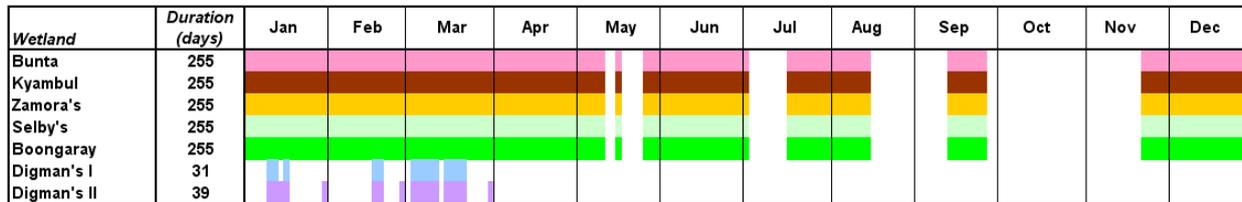
(b) Murray River

Figure 7. Timing and duration of connectivity of individual wetlands via the stream and drainage network to the (a) Tully and (b) Murray rivers during 2008 for the threshold water depth of 10 cm (from Karim *et al.*, 2010b).

Zooplankton

Zooplankton are a major component of wetland ecosystems, especially those that have slow water currents and little shading. Zooplankton (mainly small crustaceans) provide a link in the food web between phytoplankton – the main organic producer in unshaded open waters – and higher levels in the web, especially fish. They can breed and increase their numbers very rapidly in the right circumstances; conversely, their populations can crash if conditions become unfavourable. Planktonic organisms (especially water fleas) are used commonly in laboratory bioassays of water quality, and in the field they are easy to sample. They therefore provide a potentially useful monitoring tool. However, it takes substantial skill and time to identify planktonic organisms beyond the higher taxonomic categories, so discrimination among samples is hindered without substantial effort. Zooplankton abundance in the Tully-Murray floodplain lagoons showed some correlation with variables related to morphometry, water quality variables (transparency, hexazinone, dissolved oxygen) and habitat (alien plant species and alien plant cover). Some of these relationships warrant further investigation. However, because of the above constraints, currently zooplankton are not considered appropriate health indicators (Pearson *et al.*, 2010a).

Macroinvertebrates

In research completed within the MTSRF, Connolly *et al.* (2007b) have shown that macroinvertebrates are very good discriminators of subtle changes in condition in Wet Tropics streams. Their efficacy in monitoring tropical wetlands was part of the focus of research on the Tully-Murray lagoons. Invertebrate assemblages clearly reflected condition of the wetlands, but while there are significant relationships, the overall pattern was not strong. The greatest contrast in the invertebrate samples was between the habitats (litter and macrophytes), with water quality a secondary factor. This suggests that for the invertebrate assemblage at least, the lagoons under study (and at the time of the study) were in quite good condition, despite the clearing and agriculture in their immediate surroundings. It is probable that the hydrological regime, with regular flushing and through-flow, helps maintain reasonable water quality conditions. The results can be used as a benchmark for future comparisons, but cannot be considered to be representative or applied more generically for ecosystem assessments.

Birds

Birds can provide a very good indication of wetland ecosystem health. They are diverse, with a diverse relationship with wetlands, and are easy to identify and count remotely. Bird surveys were therefore included in the appraisal of the Tully-Murray wetlands (Pearson *et al.*, 2010a). However, their numbers in the lagoons were surprisingly sparse, despite the availability of abundant food for insectivorous and piscivorous species. Lagoons were not very amenable to a high abundance of wading birds, as shorelines were mostly steep, but we did expect to find good populations of diving birds (several duck species, cormorants, darter, etc.). As numbers were low, the surveys were discontinued. Absence of birds may reflect poor riparian habitat (limited perching and nesting sites), avoidance of deep waters because of crocodile threats, or other habitat/water quality issues. Water quality is considered not to be the issue, however, as birds are air breathers, and their prey (fish and macro-invertebrates) were generally abundant.

Fish

In Wet Tropics streams, Pearson *et al.* (2009) and Pusey *et al.* (2007a, 2007b) showed that fish represented condition gradients quite well, and showed moderate relationships with water quality variables, including transparency, pH, dissolved organic nitrogen, dissolved inorganic nitrogen, dissolved oxygen, conductivity, diuron, desethyl atrazine and atrazine. In the Tully-Murray wetlands, the lagoons were used as the main unit for comparing condition (rather than sites within lagoons). At this scale, which is appropriate for considering wetland monitoring and management, fish assemblages provided good indications of the environmental gradients, including general habitat quality, proximity to rivers and river mouths (Pusey *et al.*, 2007b) and connectivity. Fish require greater effort in sampling than macroinvertebrates, but the processing time of fish is much less, usually with little laboratory time required following a sampling trip. They therefore offer a distinct advantage as indicators of wetland ecosystem health (Pearson *et al.*, 2010a).

The strong associations between fish species and individual micro-habitat types within lagoons suggest that fish respond to the types of habitat disturbance typical of these floodplain wetlands. In particular, certain fish species tended to avoid areas of dense aquatic weeds including *Hymenachne* and para grass. Investigation into the factors affecting the occurrence of these weeds in floodplain lagoons indicates that the extent of invasion relates to land use and soil type, with those areas of the catchment supporting intensive agriculture (e.g. sugar cane, banana plantation), in association with alluvium soils, more likely to support *Hymenachne* (S. Januchowski, pers. comm.). Dissolved inorganic nitrogen occurs in higher concentration in wetlands draining intensive agriculture (Bainbridge *et al.*, 2009b) suggesting that agriculture has contributed to the proliferation of *Hymenachne* across the catchment. These results suggest that DIN may also be a useful indicator of anthropogenic influence on the ecological condition of floodplain water bodies. It also seems possible that low dissolved oxygen (DO) levels and/or hypoxia sufficient to discourage fish occupancy may develop within dense beds of *Hymenachne* and para grass. From these observations it is reasonable to recommend including DIN, DO, *Hymenachne* and para grass cover, and fish assemblage structure, as potentially powerful indicators of lagoon ecosystem health, and to undertake further work on these biophysical relationships and processes in lagoons of contrasting character.

The presence of early life history stages of several fish species is a revealing indicator of the degree of hydrological connectivity between saltwater and freshwater habitats. In addition, distorted patterns of fish age structure in lagoons could indicate loss of connectivity due to barriers or poor habitat/water quality in connecting channels, or altered flooding patterns due to human water use/abstraction and/or climate change. It may be possible to establish a 'connectivity disturbance gradient' embracing the full range of connectivity potential in the Tully-Murray floodplain landscape. Information contained in an audit of physical barriers (e.g. culverts, flood mitigation works, etc.) to fish passage in the Wet Tropics bioregion (Lawson *et al.* 2010) could contribute to the development of a gradient in 'connectivity disturbance' (Pearson *et al.*, 2010a).

Alien fish species can be useful indicators of reduced stream health (Kennard *et al.*, 2005; Arthington and Pearson, 2007). The presence of alien fishes is a strong indicator of initial disturbance by human activities in the broader landscape and an early warning indicator of the potential for future disturbance from increasing numbers of individuals and species (e.g. tilapia).

4.3.3 Conclusions

Region-specific monitoring systems and protocols have been developed previously in the GBR catchment (Pearson and Penridge, 1979; Arthington and Pearson, 2007), in addition to *AusRivas*, a national stream health monitoring system based on invertebrates (e.g. Smith *et al.*, 1999; Davies, 2000; Norris and Hawkins, 2000). Regional protocols are required when national systems are too broad-based to address local and regional needs (Connolly *et al.*, 2007a). Some regional protocols have been taken up for short periods, but currently there appear to be no mechanisms or resources available for systematic monitoring of ecosystem health of GBR catchments.

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

To conclude, the suite of recommended variables to describe/measure various freshwater habitats in the GBR catchments are included below.

Streams

The Catchment to Reef Joint Research Program and MTSRF research on streams in the Wet Tropics (Pearson and Penridge, 1987; Pearson *et al.*, 2003; Arthington and Pearson, 2007) and in the Mackay-Whitsunday region (Clayton and Pearson, 1996; Leonard, 2009) have greatly informed our knowledge of how these ecosystems respond to human impact. Ecosystem health of streams could be monitored by measuring a suite of variables at multiple sites along natural stream gradients as follows:

- Habitat variables, such as flow regime, flow modification, stream geomorphic characteristics, riparian extent and condition (vegetation structure, weediness, canopy cover), aquatic vegetation and alien plant infestation, excessive algal growth, leaf litter, etc.;

- Physical condition of the stream sites including: current velocity; bank stability; channel form; width; depth; sediment characteristics, including particle size and amount of detritus;

- Major water quality characteristics, including maximum and minimum values (measured through repeated 24 hour cycles) of temperature, conductivity, pH, dissolved oxygen, clarity, suspended solids, hardness, nutrients (mainly species of N and P) and short-, medium- and long-term variability in these metrics;

- Species richness of invertebrates ('species' here meaning taxa at highest level of resolution possible) and family richness of invertebrates – particularly good at the site/reach level;

- Fish species richness and assemblage composition – particularly good at the sub-catchment level; and

- Abundance and diversity of alien fish species.

Note that following further investigation, aquatic plants were considered not very useful for monitoring stream health (apart from their habitat associations with the rest of the biota) because of their high level of variability (Mackay *et al.*, 2010).

Monitoring in contrasting seasons (late wet/early dry and late dry) is required to understand extremes of conditions.

Details of the preferred monitoring strategies can be found in the above references and via http://www.rrrc.org.au/catchment-to-reef/downloads/C2R-Arthington-Aet-al-2007-Biological-Indicators-Wet-Tropics-Streams.pdf

Floodplain lagoons

Previous research on floodplain lagoons (e.g. Pearson *et al.*, 2003; Perna and Burrows, 2005), more recent studies by Pearson and others (see Pearson *et al.*, 2010a, 2009) in association with Wallace and others investigating floodplain hydrology (e.g. Wallace *et al.*, 2010b; Karim *et al.*, submitted, 2010a, 2010b, 2009a, 2009b) have greatly informed our knowledge of the nature of these ecosystems and their biota, and how they respond to human impact. We have shown here that there are neither good reference (undisturbed) sites or highly impacted sites, in terms of aquatic biota, so very strong gradients of condition are not evident in the Tully-Murray lagoons. Higher levels of disturbance were evident in the lagoons in the Herbert and Burdekin systems (Pearson *et al.*, 2003; Perna and Burrows, 2005). Nevertheless, gradients in environmental variables and significant associations of the biota with them do exist across all these systems, so we are able to outline approaches to monitoring.

Ecosystem health can be monitored by measuring a suite of variables at multiple sites and times, with some exceptions, as follows:

- Habitat variables, such as flow regime, flow modification, lagoon geomorphic characteristics (including size and depth), aquatic vegetation and alien plant infestation, riparian extent and condition, leaf litter, etc.;
- Benthic habitat (plants vs. litter) and alien plant infestation were particularly important variables for invertebrates and fish, respectively;
- Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P) and stratification; and short-, medium- and long-term variability in these metrics;
- Invertebrate diversity (mainly family levels) and assemblage structure – provide a good benchmark with regard to habitat and water quality;
- Fish species diversity and assemblage structure provide a good benchmark with regard to habitat and water quality, and to connectivity and normal movements of fish;
- Abundance and diversity of alien fish species;
- Zooplankton assemblages were not very useful for monitoring because of their low diversity and the time involved in sample processing; however, presence or absence of zooplankton could be a useful and cost-effective measure in the event of severe deterioration of lagoon condition; and
- Monitoring in contrasting seasons (late wet/early dry and late dry) is required to understand extremes of conditions, including connectivity and success of dispersal/migratory activity.

Other wetland habitats

The MTSRF research was mainly restricted to the streams of the Wet Tropics and the lagoons of Wet Tropics floodplains, but previous research on floodplain lagoons in the Burdekin and Herbert systems (e.g. Pearson *et al.*, 2003; Perna and Burrows, 2005) in conjunction with results of Pearson *et al.* (2010a), allow comment on monitoring of floodplain lagoons across the GBR catchment. For floodplain lagoons, the suite of variables of utility in monitoring is the same as indicated above for Wet Tropics lagoons. While the character of Dry and Wet Tropics systems differs greatly, differences are captured in the recommended suite of variables (including flow regime, temporal variation, etc.).

Riverine lagoons in the Dry Tropics (waterholes that remain when rivers cease to flow in the dry season) are the subject of two MTSRF-related PhD projects – one completed on water quality and algal dynamics (Preite, 2009), the other continuing on invertebrate dynamics and food webs (Blanchette, 2010). Results are not finalised but indications of metrics for ecosystem health monitoring are as follows:

- Habitat variables, such as flow regime, flow modification, lagoon geomorphic characteristics (including size and depth), aquatic vegetation and alien plant infestation, riparian extent and condition, leaf litter, etc.;

- Benthic habitat (edge, plants, sand, litter, riffle) and alien plant infestation are particularly important variables for invertebrates;

- Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P), chlorophyll and stratification; and short-, medium- and long-term variability in these metrics;

- Invertebrate diversity (mainly family levels) and assemblage structure – provide a good benchmark with regard to habitat and water quality;

- Variability among lagoons and sub-catchments requires monitoring of multiple sites; and

- Algae are time-consuming to identify and show mixed signals with regard to ecosystem health, so are not currently useful for ecosystem monitoring.

Generally, the same variables will form the basis of monitoring programs of rivers and wetlands of different character, although the study designs will need to be modified to incorporate flow regime characteristics, and physical-chemical gradients in slow-flowing, intermittent and non-linear systems, such as floodplain lagoons. It also appears that, despite lack of active management of waterways and their surrounds for improved environmental outcomes, there is substantial resilience to impacts in those systems that receive good perennial flows.

4.4 Pollutant loads

Knowledge of the loads of materials transported through waterways is critical for managers to identify pollutants of greatest concern, to quantify changes in water quality due to in-catchment actions, to set water quality targets and to assess the validity of predictive models. In many cases, the load result is one of the primary outputs of the monitoring efforts and is used as the primary reporting attribute (Brodie *et al.*, 2007a). Estimates are currently derived using a combined monitoring and modelling approach.

4.4.1 Monitoring for pollutant loads

In the Paddock to Reef Program, sub-catchment and end of catchment load water quality monitoring is led by the Queensland Government in collaboration with regional NRM bodies and monitoring providers (e.g. ACTFR, CSIRO) with a focus on sampling in major runoff events, when these exports predominantly occur. In the past, a number of monitoring programs have also been undertaken at regional levels to support NRM planning and WQIPs, with comprehensive programs established in the Burdekin (e.g. Bainbridge *et al.*, 2007), Mackay-Whitsunday (e.g. Rohde *et al.*, 2008, 2006) and Fitzroy (e.g. Packett *et al.*, 2009) catchments for event monitoring. Many aspects of these programs will continue as part of the Paddock to Reef Program. Modelling of end of catchment loads has been undertaken using SedNet and ANNEX, and an improved model, WaterCAST (see Cook *et al.*, 2009), is currently under development.

Monitoring techniques for determining the concentration of materials at sub-catchment and catchment locations are well established and generally involve the collection of grab samples around the peak of the hydrograph. Some automated instrumentation has been deployed in GBR rivers, including passive samplers for measuring pesticides and mud loggers for measuring suspended sediment. Samples can be collected at regular intervals and stored on site using refrigerated automated samplers. The MTSRF has supported development and testing of innovative load monitoring techniques including floodplain sampling described in Section 4.4.4, and sediment sampling in the Cape York catchments briefly described here.

As part of a new project jointly funded by the MTSRF and DEWHA in the Cape York region, Brooks and others have trialled the deployment of low budget sediment and recording equipment throughout the Princess Charlotte Bay catchments to obtain empirical data about sediment loads, sediment sources and the relative contributions of sediment from different geomorphic process zones within the catchment (see Brooks *et al.*, 2010). The equipment included simple integrated samplers which were deployed in all the major channels and tributaries within the catchment for the purpose of collecting representative suspended sediment samples across the whole catchment. These samples can also be used for sediment finger printing and the determination of the relative contributions of suspended sediment delivered to the GBR.

Figure 8 shows a modified version of the integrated sampler deployed for the first time as part of this study. This sampler is designed to collect a sequential deposit within the pipe protruding from the bottom of the sampler (i.e. layers of sediment that represent discrete events though the wet season). Until recently the standard practice for collecting a tracing sample was to collect bank drape material. To provide a basis for testing and comparing the integrated sampler method against the bank drape method, bank drape samples were collected in this study adjacent to all integrated samplers and the comparative results will be published in the coming months. Another innovation of the sampling method employed in this study is that temperature loggers were affixed on the integrated sampler tubes and in a high position (a tree) above flood level as a means of providing a low cost (~$20) measure of the total time that the sampler was collecting sediment and the relative timing of sampling events

for the various samplers. A series of automated cameras were also installed to provide insight into the timing and magnitude of erosion events across the wet season. Such things are otherwise difficult to determine in such remote and inaccessible areas.

Figure 8. Integrated sampler being deployed (*left*) in the Hann River. A modified version of the integrated sampler (*right*) deployed for the first time as part of this study. This sampler is designed to collect a sequential deposit within the pipe protruding from the bottom of the sampler (i.e. layers of sediment that represent discrete events through the wet season). Source: Brooks *et al.* (2010).

4.4.2 Load estimation techniques

The companion report, 'Priority Pollutants in the GBR' (Waterhouse and Brodie, in prep.)*,* provides an overview of the results of recent efforts to improve the estimates of the current end of catchment loads of sediments, nutrients and pesticides delivered to the GBR (e.g. Brodie and Waterhouse, 2009; Brodie *et al.*, 2009a, 2009c), and to identify priority areas of pollutant generation and delivery in the GBR catchments. Many of the projects were supported by the MTSRF. Advancements in knowledge for specific case studies of load estimations are described in further detail in the report, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.), and include current understanding of the influence of the Burdekin Dam on pollutant delivery and load estimations (Lewis *et al.*, 2009a) and estimations of the contribution of overbank flow to pollutant loads in the Tully catchment (Wallace *et al.*, 2009a). This report focuses on the load estimation techniques that have been developed in recent years.

There are many uncertainties and limitations associated with current load estimations, including Brodie *et al.* (2009a):

- Insufficient or ad hoc monitoring data to validate the models.
- No credible way of estimating uncertainty in load calculations. Deterministic models like SedNet/Annex provide a number but no uncertainty estimates and methods that deal with monitoring data that attempt to capture uncertainty in their load estimates only incorporate uncertainty in concentration and ignore errors in flow rates.
- Significant underestimation of the marine load during floods (Wallace *et al.*, 2009a, 2009b).
- Broad underlying assumptions about pre-European conditions are made to differentiate natural loads from anthropogenic loads to give a more realistic estimate of the change in

loads associated with human activities; resulting in uncertainty in the estimates of improvements that may be achieved through changed land management practices.

- Limited modelling capability of erosion processes in closed-forests, and of material trapping. In recent model runs it is generally calibrated to match monitoring data in that environment.

- Specific land use contributions on a regional basis are difficult to estimate and are based on broad assumptions based on proportional estimates of area. In addition, land use mapping in some catchments is dated.

- Different load algorithms can produce load estimations that vary by as much as 100% using the same monitoring data set (Kuhnert *et al.*, 2007; Lewis *et al.*, 2007a).

- Many studies do not include the whole catchment source area and, as a result, large components of the total load may be overlooked.

- Limited monitored pesticide data is available across the GBR catchment to enable regionally specific and land use specific pesticide load estimations to be developed. The ACTFR is developing an improved technique to address this.

Areas where MTSRF research has contributed to resolving these limitations are described below.

The load is defined as the amount of material, of some specified chemical or physical kind, transported by a flowing river past a specified observation location during a given period of time, often a year (reference). Loads are calculated by using the continuous flow volume of the waterway (commonly measured from hourly to daily) in combination with the concentration of a particular material to calculate the total mass exported through the sampled point of the stream (Lewis *et al.*, 2007a). This quantity is inherently difficult to assess, but two quantities are measured to inform the estimation, the *flow* of the river and the *concentration* of the transported substance. In principle, both flow and concentration are continuous functions of time. Their product is called the flux of the transported substance, also a continuous function of time. The load for a specified period is then the integral of the flux over that period of time (Venables and Harch, 2006). Estimating flow is possibly the simpler of the two in most cases, with the exception of flow estimation during flood conditions (e.g. see Wallace *et al.*, 2009a, 2009b). The main design issues connected with flow will usually be deploying the measurement resources in a cost-effective way, so that measurements in time are allocated when they are most needed. Measuring contaminant concentration, by contrast, can be much more difficult and expensive. Venables and Harch (2006) consider the statistical issues of the sampling regime (relative to the flow regime, in particular) and, more specifically, the extent to which the data record can be reliably interpolated to meet the requirements of load estimation. Brodie *et al.* (2007) and Lewis *et al.* (2007a) have considered some of the more practical issues associated with sample collection and analysis; the results of which are summarised below.

Numerous methods and software programs have been developed to calculate sediment, nutrient and other pollutant exports from waterways of the GBR catchments as described by Kuhnert *et al.* (2008) and references therein (Degens and Donohue, 2002; Fox *et al.*, 2005; Letcher *et al.*, 2002). These methods do not incorporate uncertainty in the load estimates, do not adjust appropriately for sampling biases and therefore can produce large discrepancies in the calculation of catchment loads. Such discrepancies reduce the confidence of these methods for application within the Reef Plan process, such as the setting of end-of-river load targets as well as the comparison to modelled outputs, namely the SedNet and ANNEX models (e.g. Brodie *et al.*, 2003).

Selection of a load method is strongly dependent on the concentration/stream flow data available, the hydrological characteristics of the waterway and the desired accuracy required

(Letcher *et al.*, 1999; Fox *et al.*, 2005). The optimal method(s) to calculate loads for the catchments of the GBR in the Wet and Dry Tropics of Queensland has been investigated as part of the MTSRF. The load methods (and software programs) have been rigorously assessed over different catchment areas and sampling regimes to understand the optimal sampling intervals and the best available method that will provide the desired accuracy and precision. This work was initially progressed by the CSIRO (e.g. Venables and Harch, 2006), ACTFR (Lewis *et al.*, 2007a) and Queensland government agencies (Marsh *et al.*, 2006) with continuing efforts by the CSIRO (Kuhnert and others). The findings of these studies are summarised below.

From a more practical perspective, Lewis *et al.* (2007a) investigated up to 34 different methods in three software programs to calculate loads over three different catchment areas of the GBR (a paddock in the Tully River catchment, Bowen River and Burdekin River) to examine the optimal load method and the suitable sampling frequencies over these catchment scales. The key findings of this study were:

- All software programs provided suitable load methods for the catchments of the GBR (where continuous data were available). At the time of the study, the GUMLEAF program could not be used at smaller catchment scales. The optimal methods were:
  a) **Brolga:** Linear interpolation.
  b) **Loads Tool:** Linear interpolation, inter sample mean concentration, inter sample mean concentration using mean flow.
  c) **GUMLEAF:** Flow regime stratified flow weighted mean concentration estimator (method # 19), flow regime stratified simple ratio estimator (method # 20), flow regime stratified Kendall's Ratio estimator (method # 21) and flow regime stratified Beale's Ratio estimator (method # 22).

- The minimum sampling frequencies recommended for the different catchment areas were:
  a) **Paddock scale:** Six samples evenly spaced over the hydrograph (at least two on rising limb).
  b) **Sub-catchment scale:** Daily sampling (although for catchments with very high material concentrations on the rising limb such as the Bowen, 4-5 samples per day may be required).
  c) **End-of-catchment scale:** One sample collected every two days.

The practical aspects of this study provided valuable information for the design of event load monitoring programs in the GBR catchment in terms of guiding sampling frequency. While it also provided basic guidance on the selection of a suitable load method for specific catchments, the methods do not adequately address all aspects of uncertainty which can be useful to inform future monitoring activities and reporting on the status of trends in loads (Kuhnert *et al.*, 2009), highlighting the need for further investigation of the quantification of uncertainties in load calculation techniques. Subsequently, Kuhnert and others have progressed investigations to quantify the uncertainty in loads; an overview of this work is provided below.

4.4.3 Quantifying uncertainty in load estimations

For the purpose of estimating uncertainty in quantifying pollutant loads, uncertainty is comprised of three components – measurement error, stochastic uncertainty and knowledge uncertainty, defined as:

- **Measurement error**, the uncertainty in the measured flow and concentration observed at a particular site or at different spatial locations within a site;

- **Stochastic uncertainty**, arising from the fact that not all flow and concentration data are collected; and

- **Knowledge uncertainty**, arising from our lack of understanding of the underlying hydrological processes and the ensuing choice of load estimation algorithm. A good example of this uncertainty is the largely unknown contribution of over bank floods to marine loads.

Approaches used to calculate loads range from the class of simple average based estimators, ratio estimators, infilling or interpolation approaches and the rating curve approaches. The approach developed in the MTSRF research was an extension of the regression or rating curve method, which seeks to predict concentrations based on the relationship between sampled concentrations and matched flow records for these samples (Kuhnert and Henderson 2010).

The loads methodology, LRE (Loads Regression Estimator), involves a four step process:

1) Estimation steps for flow;
2) Estimation steps for concentration;
3) Estimation of the load; and
4) Calculation of the standard error of the load.

The first step involves predicting flow at regular time intervals using a time series model such that the model captures all of the peak flows. The predicted flow is then matched to concentration sampling times to create a modelling dataset. The second step involves the characterisation of the relationship between concentration and flow through a generalised additive model (GAM) that incorporates all important covariates in an attempt to capture the underlying hydrological processes and minimise knowledge uncertainty. Predictions are made at each regularised flow value ensuring that predictions are capped at the maximum concentration observed in the dataset. Predicting at regular time intervals is the key to accounting for bias in the sampling process. An estimate of the load in the third step is obtained by multiplying the predicted concentration and predicted flow, summing the calculation and incorporating a unit-conversion constant for time interval used. Standard errors of the load are then computed during the fourth step of this process, which incorporate errors in the concentration and flow samples with the latter incorporating measurement error and errors due to the spatial location of sampling sites.

The generalised rating curve approach is novel as it seeks to represent a number of important system processes for GBR catchments to account for expected or implied system behaviours:

- **First Flush**, the first significant channeled flow in a water year accompanied by high concentrations (represented as a percentile of flow and used in the calculation of other system processes);

- **Rising/Falling Limb**, which allows higher or lower concentrations on the rising limb when runoff energies are higher and sediment supply may also be higher. This is usually represented at shorter time-scales than exhaustion, which is parameterised for between-event variations. This covariate is based on the flush (process 1) defined for that period;

- **Exhaustion**, representing the limited supply of sediments and nutrients due to previous events (represented by a discounted flow term);

- **Hysteresis**, representing complex interactions between flow and concentration with strong historical effects and dependence captured by non-linear terms for flow and incorporating hydrological processes 1-3; and

- **Overbank Flow**, described as flow that goes over bank in flood events which can be quantified using hydro-dynamic models. This process is currently being investigated by Wallace *et al.* (2010c).

The methodologies were applied to two real long-term monitoring datasets: the Burdekin River at Inkerman Bridge and the Tully River at Euramo. See Kuhnert and Henderson (2010) for a detailed summary of the analyses for each site.

The LRE package was applied to these datasets to estimate loads and uncertainties for each water year represented. The results are shown in Figures 9 and 10 for the Burdekin and Tully rivers respectively and can be summarised as follows:

- **Burdekin**
  – Annual loads and mean annual concentrations were estimated for the Inkerman Bridge site for 36 water years using the LRE methodology. Summaries of the data indicated considerable bias in the concentration sampling with no bias in flow samples due to the regular sampling intervals (hourly).
  – A model was fit to 824 concentration samples with linear and quadratic terms for flow (a seasonal term) and smooth terms for the discounted flow and trend. Results showed a reasonable fit with 69.9% of the variance explained. A seasonal term fit in the model showed increases in TSS concentration during the wet months (October to April) and decreases during the drier months of the year (May to September).
  – Average mean concentrations were higher in some years compared to others. Further investigation revealed cyclones that had passed through the Bowen sub-catchment of the Burdekin. Inclusion of terms that reflect these events in the model may help explain increases in concentration for this catchment.

- **Tully**
  – Data for the Tully River at Euramo Bridge spanned 35 years and were used in an analysis using the LRE package to estimate loads with uncertainties. Unlike the Burdekin River, flow for the Tully was collected at irregular time intervals ranging from 0 hours to 43.91 days with a mean of 1.015 hours and a median of 2.24 days. Summary statistics showed substantial bias in the concentration in addition to the biased sampling of the flow.
  – A model fitted to 489 concentration samples highlighted linear and quadratic terms for flow, a seasonal term, a rising/falling limb term and a discounted flow term that was important for predicting concentration and explained 74.2% of the variation in the data. The seasonal term indicates decreasing concentrations from November through to the end of June and an increase from July through to the end of October. The rising/falling limb term fitted in the model was significant and indicates an increase in concentration (approximately 2.3 times) on the rise of an event, compared to on the flat. A decrease in concentration on the fall is noted, although it is not significant.

–   Average mean concentrations show a large load (and uncertainty estimate) occurring in 1994/95. Apart from this estimate, the average mean concentrations predicted for all remaining water years exhibit a cyclic behaviour, where approximately every ten years the load appears to increase. Further investigation into the behaviour of these estimates and whether, like the Burdekin, certain climatic events have contributed to this are required.

The methodologies were also applied to end of river sites in the GBR through a project funded by the Queensland Department of the Premier and Cabinet (DPC) where monitoring data were available and representative of the river system at each location (Kroon *et al.*, 2010). Sites investigated included Normanby, Barron, Johnstone, Tully, Herbert, Haughton, Burdekin and Pioneer rivers for nine pollutants consisting of total suspended sediment (TSS), total nitrogen (TN), dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON), particulate nitrogen (PN), total phosphorus (TP), dissolved inorganic phosphorus (DIP), dissolved organic phosphorus (DOP) and particulate phosphorus (PP). Total annual estimates of pollutants with uncertainties were calculated for each site and then converted to mean-annual loads (referred to as 'long-term' loads), which can be likened to a long-term average produced from SedNet/Annex. Estimates of uncertainty were also evaluated. Details of these calculations can be found in Kroon *et al.* (2010).
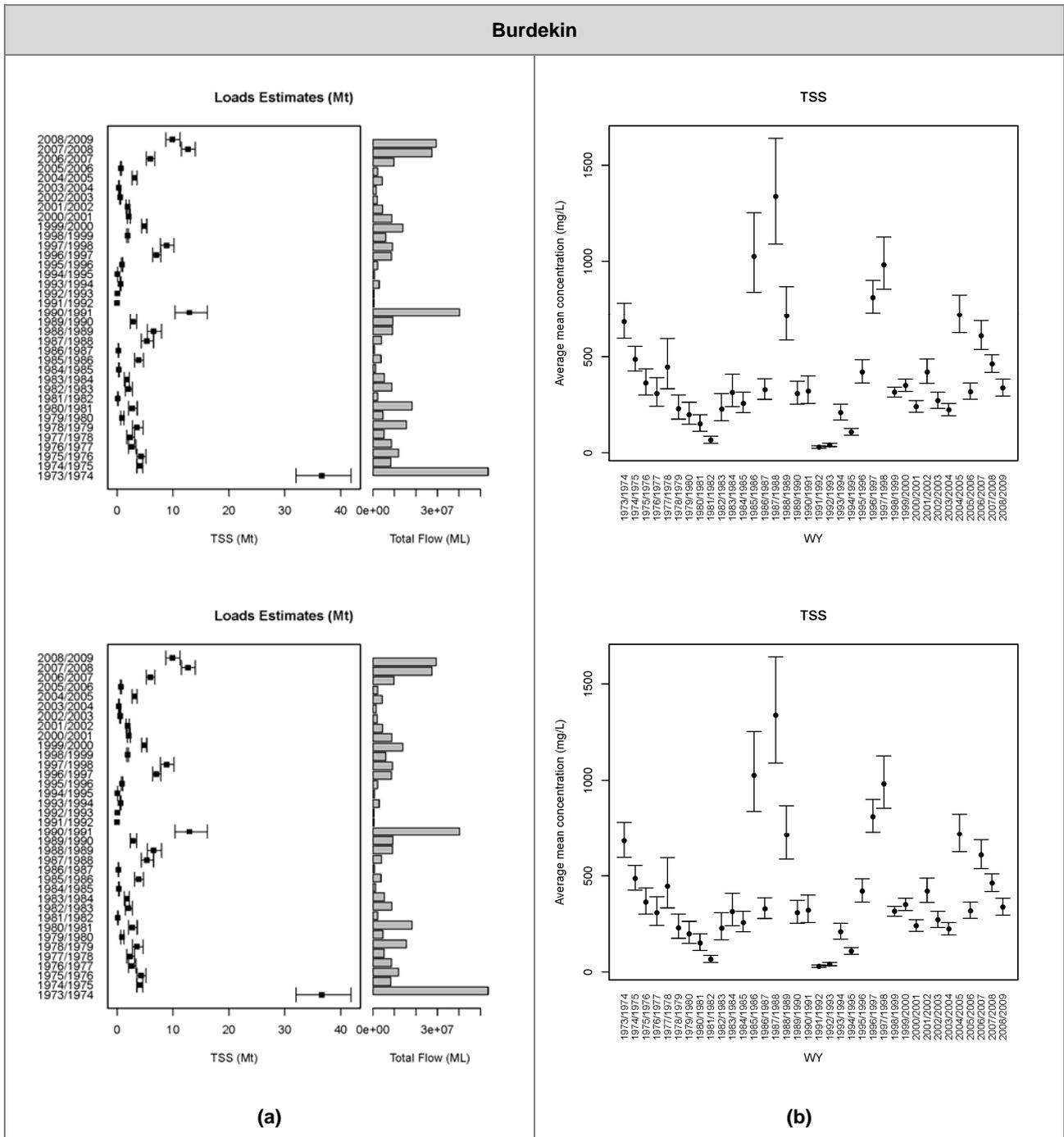
Figure 9. Plots for the Burdekin site showing (a) the estimated TSS load (Mt) and 80% confidence intervals for each water year accompanied by the total volume of flow (ML), and (b) the average mean concentration (mg/L) for each water year (from Kuhnert and Henderson, 2010).
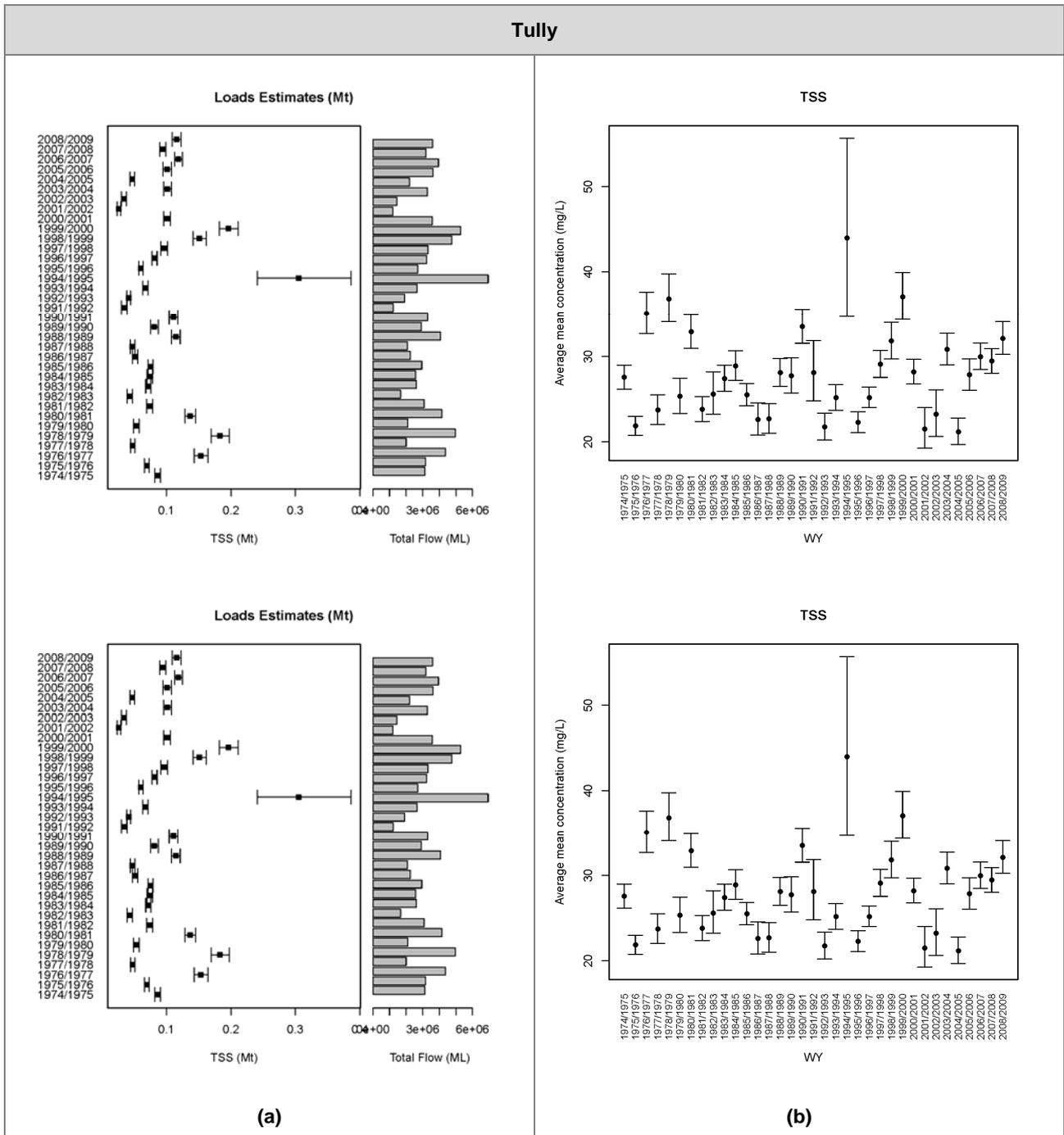
Figure 10. Plots for the Tully site showing (a) the estimated TSS load (Mt) and 80% confidence intervals for each water year accompanied by the total volume of flow (ML), and (b) the average mean concentration (mg/L) for each water year (from Kuhnert and Henderson, 2010).

The LRE methodology is a significant advancement on previous approaches to loads estimation using monitoring data for several reasons, outlined below:

- The methodology presented here is a regression based methodology that incorporates terms to mimic key hydrological processes operating in a river system. It is also general enough to incorporate a suite of additional variables (e.g. sources, management interventions, structures such as a dam) into the model depending on the type of river being analysed.

- The nature of the additive model allows the inclusion of smooth, flexible terms which can help to explain large sources of variability in the data.

- Unlike previous static approaches such as the average, ratio and interpolation estimators, LRE can make use of the existing data and borrowing strength across years when characterising the relationship between flow and concentration which is used for prediction and loads estimation. This may mean that reduced monitoring is required for future years once the relationships are well characterised.

- Provided that the data are representative of the river system, this method has the ability to predict where there are gaps in concentration sampling and provide load estimates where no concentration data were collected. We do however caution about predicting outside the range of the data because it is possible that relationships between characteristics of flow and concentration may change over time, and that applying it may deliver poor load estimates for years outside the range.

- Adjusting for bias and accounting for uncertainty in flow and concentration is explicitly captured in the LRE methodology.

    - Bias is taken into account by predicting concentration at regular flow values and calculating a load.

    - Uncertainty relating to concentration is captured through the generalised additive model used to characterise the hydrological system, while errors in flow rates are captured through two coefficients of variation estimates that can be set in the model. The first captures measurement errors in sampled flow while the second coefficient of variation expresses the error in the spatial positioning of the gauge in the river.

4.4.4 Incorporating near coastal areas

Current load estimates are largely based on measurements usually collected at the lowest gauging station in the catchment; this location may exclude coastal areas downstream of the station, some of which may incorporate large areas of cropping land uses, and any over-bank flow that is not captured by standard river gauges. The problem of excluding near-coastal areas in estimations of total catchment load is much larger for monitoring than for modelling load estimates (Brodie *et al.*, 2009a). Two prominent examples of this issue arising in the GBR catchment are in Burdekin and Tully basins.

In the Burdekin basin, loads are often estimated through modelling for the mouth of the Burdekin River and monitoring of loads is carried out at the sampling site near Home Hill, also near the river mouth. However, most of the drainage from sugarcane cultivation in the Burdekin basin doesn't flow past this sampling site in the main river. Pollutants from sugarcane land discharge to the GBR via small streams, e.g. Barratta Creek, and via groundwater discharge. Any loads modelled or monitored at Home Hill in the main river will not include these discharges and an extra estimation step is required to measure the total load, including the sugarcane load, from the Burdekin basin to the GBR (Brodie and Waterhouse, 2009; Brodie and Bainbridge, 2008). Methods to estimate this extra load are being developed and future estimates of GBR loads will include this factor (e.g. Kroon *et al.*, 2010). There are many other similar examples of this in other river basins, e.g. Burnett River.

In the past, Tully River loads have been estimated from monitoring at the gauging station in the river channel at Euramo (approximately fifteen kilometres upstream of the coast). However, in high-flow events, the Tully and Murray Rivers break their banks and the floodwaters merge and flow to the ocean as a large sheet of water many kilometres wide. During these conditions the river gauges do not record the total catchment discharge very well. For example, Wallace *et al.* (2009b) showed that during the thirteen floods between 2006 and 2008, the Tully River gauge at Euramo only recorded 36-88% of the flood discharge and the Upper Murray gauge only recorded 11-27% of the flood discharge. Furthermore, current ocean sediment and nutrient loads are based on concentrations measured within the rivers, yet until the MTSRF project was initiated, the sediment and nutrient concentrations are in over bank flood waters were not known. Wallace *et al.* (2009a, 2009b, 2010a) present new estimates of flood discharge that include over bank flows combined with direct measurements of sediment and nutrient concentrations in flood waters to calculate the loads of sediment and nutrient delivered to the ocean. Although absolute concentrations of sediment and nutrient were quite low, the large volume of water discharged during floods means that they make a large contribution (30-50%) to the marine load (Table 4). Furthermore, by not accounting for flood flows correctly, previous estimates of the annual average discharge are 15% too low and annual loads of nitrogen and phosphorus are 47% and 32% too low respectively. The size of this underestimate in any year will depend on the number and size of over bank flood events in that year. This will make the monitoring of any underlying trends in marine loads difficult unless it is possible to remove inter-annual variability in flood load contribution. As sediments may be source limited, accounting for flood flows simply dilutes their concentration and the resulting annual average load is similar to that previously estimated.

Table 4. Long-term (1972-2008) annual average sediment and nutrient loads leaving the Tully and Murray catchments. Total loads are separated into those occurring while flow is in-bank and over-bank (i.e. during flooding). For comparison, the annual average loads from all of the published studies in the Tully and Murray catchments are also shown. Source: Wallace et al. (2009b).

	All studies (tonnes)			Wallace *et al.* (2009b) (tonnes)		
	TN	**TP**	**TSS**	**TN**	**TP**	**TSS**
In-bank	1129	114	76097	1129	114	76097
Over-bank flood	543	55	36626	1322	109	38842
Total	**1672**	**169**	**112723**	**2450**	**223**	**114939**

Another important finding was the composition of the flood waters compared to river waters, showing that concentrations of dissolved organic nitrogen are higher than dissolved inorganic nitrogen in flood waters which is the opposite of river water (Wallace *et al.*, 2009a, 2009b, 2010a). This has implications for load based estimations in this catchment and those with similar over bank conditions and the type of management practices that may be adopted to reduce nutrient delivery to the GBR.

Given the above, in catchments that experience over-bank flow in flood periods, monitoring of marine loads will require a significant number of samples of both river and flood flows (in time and space) – otherwise the large uncertainties in mean loads may be misleading and it may be difficult to detect any load reduction trends (Wallace *et al.*, 2010a). Assessments of which GBR rivers may contain significant over-bank flood loads are now being made by Wallace *et al.* (2010c).

The above findings were determined through the development and application of innovative techniques to measure pollutant loads on the floodplain in flood conditions (see Hawdon *et al.*, 2007). One of the lessons learnt from preliminary manual sampling during the floods that followed Tropical Cyclone *Larry* in March 2006 was the difficulty in obtaining access to the catchments during floods. During the first few days of these events roads are often impassable and it can be impossible to reach the centre of the flooded areas from either north or south of the catchments. Flood water sampling systems were therefore developed that could automatically collect water samples during the early parts of a flood. These samples are then collected manually several days later when access can be gained.

The most suitable device for water quality sampling is a fully automatic and refrigerated sampler (ISCO Avalanche auto sampler). However, it would be prohibitively expensive to deploy many of these across the floodplain, so a hybrid system was designed that included three automatic water quality monitoring systems that measure sediment and nutrient concentrations in over-bank flood waters. The systems include a fully automated monitoring station with a refrigerated auto-sampler and telemetry, two programmable temporal water quality samplers and eight passive rising stage samplers. The components, construction and application of each of the above systems is described in Hawdon *et al.* (2007), in conjunction with the logic of each of the automated systems and sampling sequence associated with each system.

4.4.5 Pesticide load estimation techniques

Lewis and others (see Brodie *et al.*, 2009c) have established a method to estimate end-of-catchment pesticide loads for the GBR. Runoff coefficients were established for the six key herbicides designed to inhibit photosystem II in plants and commonly detected in the GBR lagoon, including diuron, atrazine, hexazinone, ametryn, simazine and tebuthiuron (Brodie *et al.*, 2009c). These coefficients were developed based on the loads calculated for the Haughton River, Barratta Creek, Pioneer River, Sandy Creek, O'Connell River and Fitzroy River and published in Lewis *et al.* (2009b) and Packett *et al.* (2009). The event mean concentrations (EMC) were calculated from these loads and an average EMC for each stream was calculated where multiple years were monitored. The 'average load' was then calculated using the 'average discharge' from each of the monitored streams determined by a SedNet model run (Brodie *et al.*, 2003). The upstream land use (in hectares, using Queensland land use mapping (QLUMP 1999 data) for the sampled point of each stream was then established and major land uses included. These were forest, grazing, sugar, other crops and other (includes urban, water storages, etc). The average herbicide runoff in kg per hectare was then calculated for sugar areas assuming that all loads of diuron, atrazine, hexazinone and ametryn from the Haughton River, Barratta Creek, Pioneer River, Sandy Creek and O'Connell River are sourced to the sugar industry. This assumption is reasonable given that: (1) sugar is the predominant industry within these regions, and (2) these herbicides are widely used in sugar and studies have shown a direct relationship between sugar area and the concentration of these herbicides in streams (e.g. Bainbridge *et al.*, 2009a; Lewis *et al.*, 2009b).

The mean of the runoff coefficients for each stream was then taken to produce an average coefficient for sugar areas in the GBR catchments; this coefficient was then used to estimate the loads of these herbicides. Similarly, runoff coefficients for atrazine, diuron and simazine in cropping areas were generated using the load data from the Fitzroy River and assuming that these herbicides in this river are only sourced to cropping area lands. This coefficient was then applied to the 'other crops' land use to estimate atrazine, diuron and simazine loads. Since tebuthiuron is only sourced to the grazing industry, a runoff coefficient for tebuthiuron in grazing lands was developed using the mean kilogram per hectare load data from the Fitzroy River, Haughton River and O'Connell River. As there were considerable differences in the individual kilogram per hectare calculations between the Fitzroy River and

the Haughton and O'Connell Rivers, the calculation for the Fitzroy River was used exclusively for dryland grazing lands (including the Burdekin and Fitzroy Rivers).

These methods have been applied in establishing the baseline pesticide loads for the GBR catchments as part of the Paddock to Reef Program (Kroon *et al.*, 2010), in guiding investment through the Reef Rescue Program and in the development of priority areas for the introduction of regulations for agricultural areas in Queensland (Brodie and Waterhouse, 2009; Brodie *et al.*, 2009c).

4.4.6 Conclusions

The recent development of baseline loads for the Paddock to Reef Program (see Kroon *et al.*, 2010) has highlighted several issues relating to the calculation of baseline pollutant load estimates for the GBR. At present, there are two primary approaches for estimating a pollutant load:

1) A (deterministic) process-based model (e.g. SedNet and the soon to be released WaterCAST) that incorporates mapped information about different sources of erosion and takes into account the hydrology and contaminant transport characteristics of the system. This information is used to route the pollutants through a river network and to estimate a load; and

2) A statistical modelling framework (LRE) that makes use of monitoring data collected at a site within a catchment over a specified time frame.

The decision as to which model to use is largely subjective and depends on the resolution and representativeness of the data captured and how well the process model is believed to mimic the underlying hydrological processes and variability of the system. Where the monitoring data is representative of the river system, statistical approaches tend to be applied as in Kroon *et al.* (2010); when monitoring data is sparse or unavailable, process-based models are typically used. As a result, there is currently a mixture of the two types of models applied throughout the GBR catchments to estimate pollutant loads and inform a baseline in the Paddock to Reef Program (subjective analysis). In addition, process based models are calibrated using monitoring data that are used as a means for calculating loads, ignoring uncertainty in the model structure as well as on the data that are used for calibration purposes. This mismatch of methods results in load estimates developed for different catchments, at different spatial scales, with different sources of error, making it difficult to monitor and track change (if any) through time and in space – an outcome which is considered a high priority for Reef Rescue Research and Development investment. To ensure that the resultant load estimates are beyond reproach, it is essential that all sources of uncertainty (parameter, model and data) associated with load estimates are propagated through the catchment models, resulting in transparent, objective and repeatable estimates of end-of-catchment loads.

The use of process-based models to estimate loads for paddock, catchment and marine components of the GBR is proposed in the design of the Paddock to Reef Program. The difficulty in relying solely on models (with parameters calibrated using monitoring data) is that monitoring data are unavailable in some parts of the GBR. This may result in an unrealistic and biased load estimate for these areas since the model is not calibrated to actual values. Further development of a model that assimilates both modelled and monitoring data are required to provide an objective and repeatable analysis for load estimation that accounts for the uncertainty in both the monitoring data and in the modelled estimates.

Another key aspect of pollutant load estimation from a management perspective is detecting changes over time, particularly in response to catchment initiatives designed to reduce loads.

This is difficult because GBR pollutant loads can exhibit substantial inter-annual variability, driven by large variations in rainfall. In making the comparison between years we typically consider loads under 'average' conditions. This is what SedNet does as it is a long term average. For the Paddock to Reef Program baseline pollutant load estimates (Kroon *et al.*, 2010), loads were averaged in some way over the different years that monitoring data were available. The LRE method currently provides load estimates for each year, drawing upon all available monitoring data to characterise relationships between flow and concentration. When data for a new year comes in the model is updated and used to predict that year. This means it may take some time for new data to update the relationship enough to show up in different load estimates. Modifications are, however, possible to improve the ability to detect changes over time (e.g. allowing time-varying relationships, comparing estimates derived from data over different time ranges). This may require an update to how 'average loads' are calculated if the focus is on detecting change rather than one-off baseline estimates.

The implications of the flood water quality studies in the Tully and Murray catchments, and potentially for other GBR catchments, are as follows:

1) Over-bank floods can make a large contribution to the marine load of sediment and nutrients and much of this load may not be recorded by standard river gauges.

2) In GBR catchments where floods are a significant proportion of the annual flow, current marine load estimates of sediment and nutrients (based on gauged flows, measured river concentrations and modelling) are probably too low, by significant amounts, depending on estimation method and constituent.

3) The size of this underestimate in any year will depend on the number and size of over-bank flood events in that year. This will make the monitoring of any underlying trends in ocean loads difficult unless it is possible to remove inter-annual variability.

4) Monitoring of marine loads will take a significant number of samples of both river and flood flows (in time and space) – otherwise the large uncertainties in mean loads may be misleading and it may be difficult to detect any load reduction trends.

5) The cause of the above underestimate in loads is mainly due to the poor recording of flood (over-bank) discharges by river gauges, but also to differences in flood water and river water quality concentrations.

6) Flood waters can carry more dissolved organic nitrogen (DON) than dissolved inorganic nitrogen (DIN) and this is the opposite of their concentrations in river water. Consequently DON loads to the ocean may be much higher than those previously estimated from riverine data.

7) Land management actions that focus on farm interventions in agriculture will potentially reduce DIN loads.

8) Reductions in DON (and sediment) loads that arise outside the floodplain require different interventions to those used in agriculture to reduce DIN; e.g. measures that slow down and reduce drainage and the introduction and/or rehabilitation of riparian zones and wetlands.

The inaugural flood water quality data collected in the Tully and Murray catchments has demonstrated the importance of obtaining observations from the key processes that control the marine loads of concern. In the Wet Tropics catchments studied, in addition to chanelised flow, over-bank flooding is a primary material transport mechanism and it is very difficult to adequately capture this process in monitoring and/or modelling schemes that are entirely river based. There is therefore a clear need to obtain estimates of the contribution that floods make to marine loads in other GBR catchments.

4.5 Estuarine and marine ecosystem health and response

As outlined in the 2008 Scientific Consensus Statement on Water Quality in the Great Barrier Reef (Brodie *et al.*, 2008a, 2008b), understanding of the effects of land-sourced contaminants on GBR species and ecosystems has expanded enormously in the period since 2003. However, the size of the system and its temporal variability mean that 'representative' monitoring and measurement of conditions in the water column and of ecosystem condition is difficult. The impacts of water quality on corals have been demonstrated through laboratory and field studies and data synthesis and integration has enabled the development of trigger values/thresholds of corals to water quality parameters. Knowledge related to the impacts of water quality on seagrasses has been synthesised, and efforts to understand the synergistic effects of multiple stressors on corals and seagrasses have commenced. The findings of the MTSRF research have contributed significantly to this progress, and are outlined below.

4.5.1 System understanding and thresholds of concern

Development of suitable targets for GBR water quality ecosystem health requires an intricate knowledge of the relationships between degraded water quality conditions and ecosystem health. Through several years of research, strong links between coral reef health and water quality conditions have been shown at local scales (reviewed in Fabricius, 2005; Cooper *et al.*, 2008), at regional scales (Devantier *et al.*, 2006; van Woesik *et al.*, 1999; Fabricius *et al.*, 2005), and recently at a GBR-wide scale (De'ath and Fabricius, 2008, 2010; reviewed by Lewis *et al.*, 2009b). Predicted ecological effects of poor water quality (elevated delivery and/or concentrations of suspended sediment, nutrients or pesticides) on corals and coral reproduction have been a focus area for the MTSRF, building on work from the Catchment to Reef Joint Research Program. The most recent and comprehensive review of this work is provided in Fabricius (2010), and covers how nutrient enrichment directly and indirectly affects corals and other reef-associated organisms, how the ecological balance of coral reef ecosystems changes with nutrient enrichment, light loss and sedimentation, and factors influencing the susceptibility of reefs to eutrophication. A qualitative conceptual model of this knowledge is also presented to describe these relationships.

Evidence of declining seagrass health due to reduced light availability, increased sediment deposition and elevated nutrient loads (see Waycott and McKenzie (2010) for an overview of these relationships) are also observed. As seagrasses, along with other coastal habitats, buffer the influence of terrestrial impacts, ongoing declining water quality that affects seagrasses will lead to additional impacts on coral reefs (Grech *et al.*, 2010; Waycott and McKenzie, 2010).The vulnerability of seagrasses in the GBR to climate change is summarised in Waycott *et al.* (2007).

Reef ecosystem health

Abundances of a range of reef associated organisms have been shown to change along water quality gradients. Figure 11 summarises the results of a review of existing reef studies from around the world to identify the main effects of nutrient and sediment related parameters on key groups of coral reef organisms. The data suggest that nutrient enrichment can lead to macroalgal dominance if light levels are sufficient, but lead to dominance by heterotrophic filter feeders if light becomes a limiting factor for macroalgae (Johannes *et al.*, 1983; Birkeland, 1988). It also shows that crustose coralline algae, which are essential settlement substratum for coral larvae, are negatively related to sedimentation (Fabricius and De'ath, 2001), as later confirmed by laboratory experiments (Harrington *et al.*, 2005).

While pollution effects on coral reefs at local scales are well understood, links at regional scales between increasing sediment and nutrient loads in rivers, and the broad-scale

degradation of coral reefs, have been more difficult to demonstrate (Fabricius and De'ath, 2004). This is due to a lack of large-scale historic data and the confounding effects of other disturbances such as bleaching, cyclones, fishing pressure and outbreaks of the coral eating Crown-of-Thorns starfish, and is further complicated by the naturally high variability in monsoonal river flood events. However more recently, relationships between data sets of water quality, and macroalgal cover and the richness of hard corals and phototrophic and heterotrophic octocorals, have been investigated at a GBR-wide scale (De'ath and Fabricius, 2008, 2010).

	DIN	POM*	Light reduction	Sedimentation
Crustose coralline algae	↓			⬇
Bioeroders	↑	⬆		↓
Macroalgae	⬆	↑	⬇	↓
Heterotrophic filter feeders		⬆	⬆	↓
Coral diseases	↑			↑
Coral predators		⬆		

Figure 11. Effects of the four main parameters of terrestrial runoff on organisms that interact with corals. High abundances of crustose coralline algae as settlement substrata promote coral populations, whereas high abundances of the other groups are assumed to negatively affect coral populations. The arrows indicate the relative strength and direction of the response (arrows pointing up or down = increasing or decreasing, thick arrow = strong effect, thin arrow = weak effect); empty cells indicate that insufficient data are available. POM = Particulate Organic Matter. Source: Fabricius (2005).

While pollution effects on coral reefs at local scales are well understood, links at regional scales between increasing sediment and nutrient loads in rivers, and the broad-scale degradation of coral reefs, have been more difficult to demonstrate (Fabricius and De'ath, 2004). This is due to a lack of large-scale historic data and the confounding effects of other disturbances such as bleaching, cyclones, fishing pressure and outbreaks of the coral eating COTS, and is further complicated by the naturally high variability in monsoonal river flood events. However more recently, relationships between data sets of water quality, and macroalgal cover and the richness of hard corals and phototrophic and heterotrophic octocorals have been investigated at a GBR-wide scale (De'ath and Fabricius, 2008, 2010).

The methods used to undertake the analysis are outlined in De'ath and Fabricius, (2010). Water clarity (Secchi disk depth) and water column chlorophyll were used as measures of water quality (Figure 12). The relationships between water quality on four benthic parameters – macroalgal cover, species richness of hard corals, and generic richness of phototrophic and heterotrophic octocorals (soft corals and sea fans) (Figure 13) were considered. The analysis comprised three stages (described further in De'ath and Fabricius, 2010).

1) **Spatial analysis of water clarity and chlorophyll, and four benthic parameters.** Data from each of the six parameters were reef-averaged, and the spatial distribution of each was modelled. The spatial predictors were relative distance across and along the GBR shelf (hereafter across and along), as opposed to the traditional latitude/longitude (Figure 12a).

2) **Modelling each of the four benthic parameters as functions of water clarity, chlorophyll, and across and along.** Since water clarity and chlorophyll were sampled at different sites than the benthic parameters, their values at the benthic sites were predicted using the spatial models of stage one.

3) **Guideline values for water clarity and chlorophyll conditions were determined.** Based on mean values of the coastal and inshore waters of the far northern GBR and the outputs from the models relating the four biotic responses to water clarity, chlorophyll, and across and along. All reefs of the GBR exceeding the guidelines were identified, and the boosted regression tree models were used to predict potential changes in the benthic parameters on these reefs should water clarity and chlorophyll concentrations be improved to meet the guidelines.
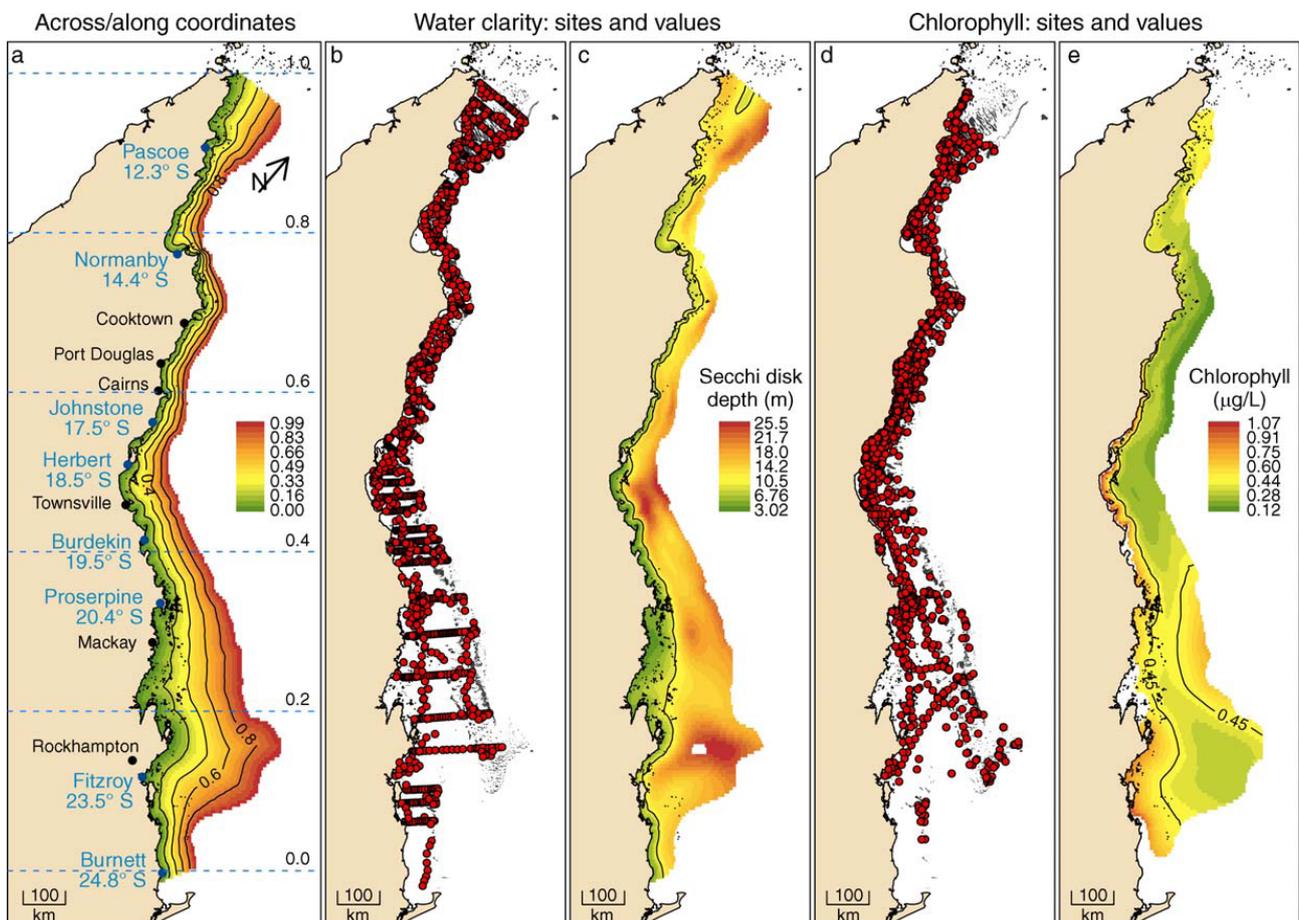
Figure 12. Maps of the GBR illustrating (a) the across/along coordinate system, the spatial distribution of (b) water clarity stations (N = 4067), (c) water clarity values (Secchi disk depth), and (d) chlorophyll stations (N = 2058), and (e) chlorophyll values. In (a), the black solid lines are contours for the values of across, the blue dashed horizontal lines are contours for the values of along, and the names in blue indicate rivers, with the latitude of their mouths added for reference. Maps of (c) water clarity and (e) chlorophyll values show the critical 10m and 0.45µg/L contours, respectively. Unreliable predictions (>3 mean standard errors of predicted values) were excluded from plots. Maps of the SE of predictions are shown in the Appendix of the source. Source: De'ath and Fabricius (2010).
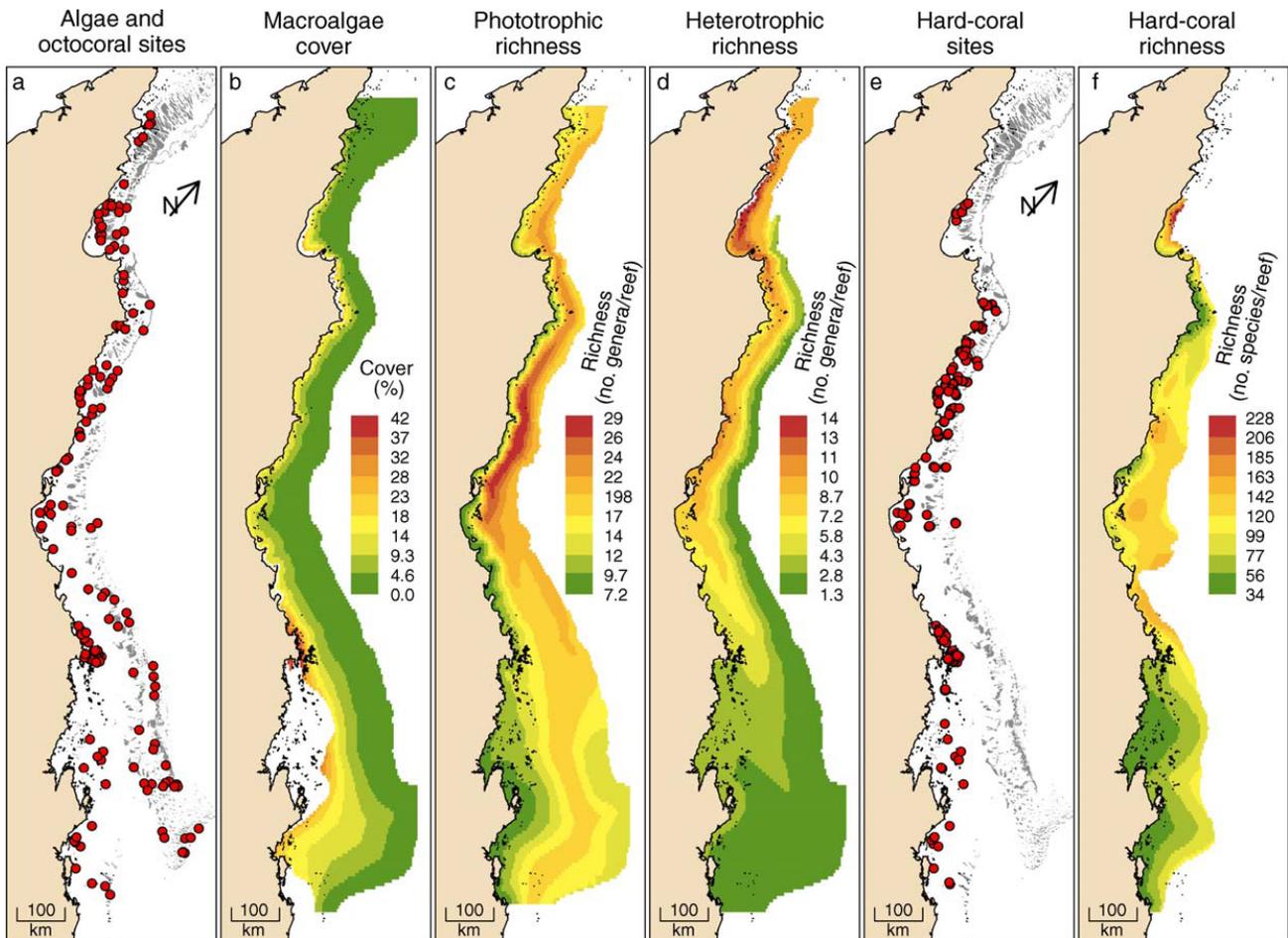
Figure 13. Maps of the GBR illustrating the spatial distribution of (a) macroalgae and octocoral sites (N = 150), (b) macroalgal cover, (c) phototrophic and (d) heterotrophic octocoral richness, (e) hard-coral sites (N = 110), and (f) hard-coral richness. Unreliable predictions (>3 mean SEs of predicted values) were excluded from plots. Maps of the SE of predictions are shown in the Appendix of the source. Source: De'ath and Fabricius (2010).

The study showed that the four biotic indicators chosen are significantly related to GBR water quality. Macroalgae increased and hard coral richness and the richness of phototrophic octocorals declined with increasing turbidity and chlorophyll, after cross-shelf and long-shore effects were statistically removed (Figure 14). Heterotrophic octocorals slightly benefited from high turbidity. Mean annual values of >10 m Secchi depth and <0.45 µg L^{-1} chlorophyll were associated with low macroalgal cover and high richness of phototrophic octocorals and hard corals. The study suggested these values to be useful water quality guideline values. These guidelines are presently exceeded on 650 of the 2800 gazetted reefs of the GBR. The models showed that compliance with these guideline values by, for example, minimising agricultural runoff would likely reduce macroalgal cover by ~50% and increase hard coral and octocoral richness by 40% and 70%, respectively, on these 650 reefs.
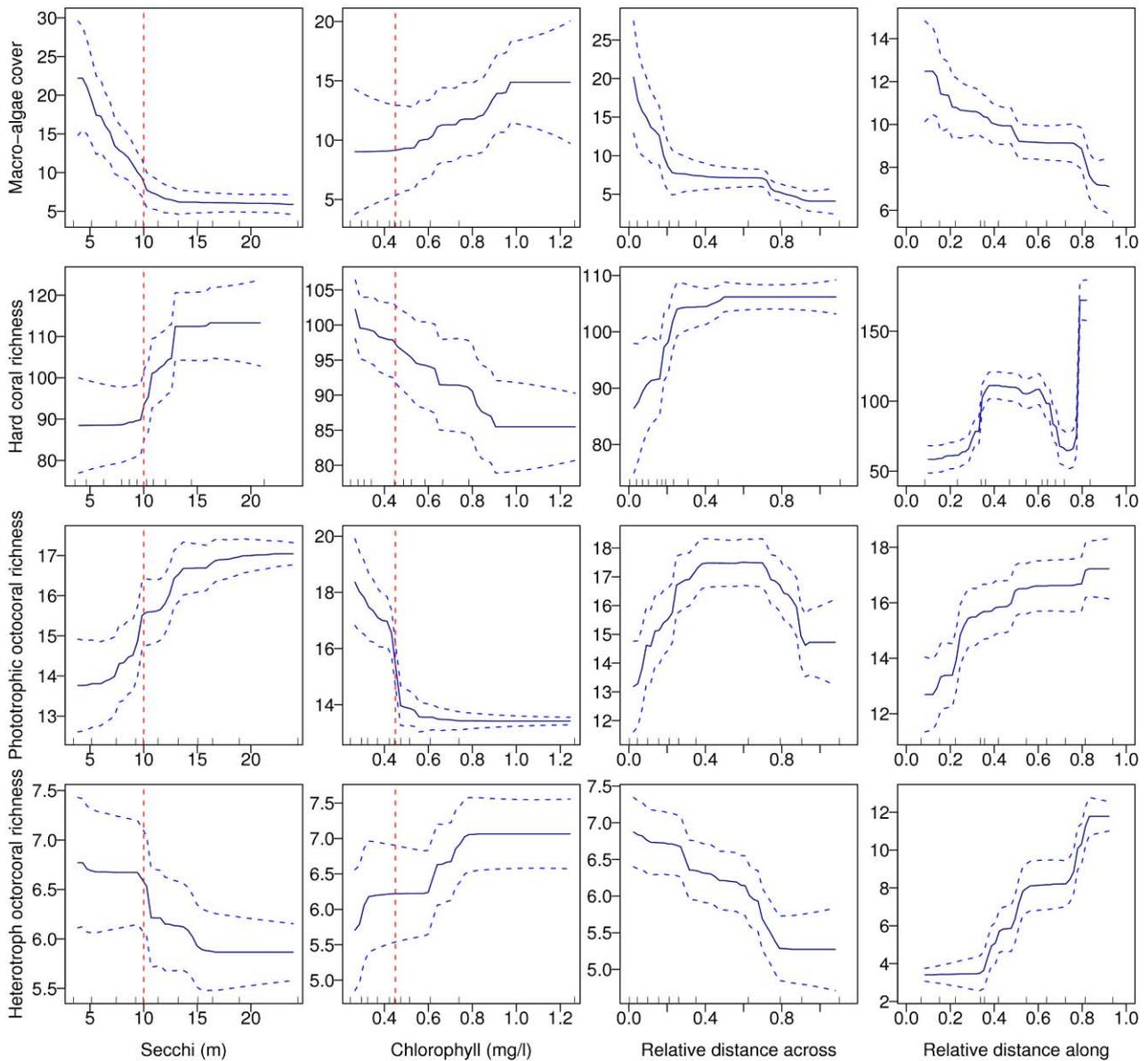
Figure 14. Relationship of macroalgal cover, and the taxonomic richness of hard corals, phototrophic and heterotrophic octocorals (soft corals and sea fans with and without zooxanthellae, respectively), along gradients in water clarity (measured as Secchi disk depth) and chlorophyll, while also controlling for relative distance across and along the shelf. Substantial increases in macroalgal cover and losses in coral biodiversity are being observed at <10 m Secchi disk depth, and >0.45 μg L^{-1} chlorophyll. The red lines show the proposed water quality guideline values (10 m Secchi disk depth, and 0.45 μg L^{-1} chlorophyll). Source: De'ath and Fabricius (2008).

The required changes in coastal and inner shelf chlorophyll and water clarity were calculated for each of the NRM regions (De'ath and Fabricius, 2008). In coastal waters, reductions in mean annual chlorophyll by 22 to 63%, and increases in water clarity by 56 to 170% will be necessary to re-establish highly diverse coral communities and reduce abundances of macroalgae. The required changes would be greatest north of the mouths of the Burnett, Fitzroy, Burdekin, Herbert, Tully and Johnstone Rivers. In inner-shelf waters, water clarity is close to the proposed guideline trigger value in all regions, while chlorophyll would need to be reduced by 46% and 8% on inner-shelf reefs of the Burnett Mary and the Burdekin

regions, respectively, to allow reef biodiversity to recover. Given the strong correlations between chlorophyll, SS, PN and PP, a reduction in chlorophyll and Secchi depth may be achieved by efforts in reducing loads of SS, PN and PP in rivers.

In coastal reefs of all regions other than Cape York, macroalgal cover would likely be reduced to about half of current values if water clarity and chlorophyll were improved simultaneously. Changes in macroalgal cover were more strongly related to water clarity than to chlorophyll. Due to the natural north-south gradient in macroalgal cover, macroalgal cover would still be naturally higher in the three southern regions compared to the northern regions after water quality improvements were implemented. Hard coral richness on coastal reefs in the Burnett Mary, Fitzroy and Wet Tropics would likely increase by 44-47% compared to present-day values, and in the Mackay-Whitsundays and Burdekin by about 30%. On inner shelf reefs, hard coral richness would still increase by 20-25% in the Fitzroy and Mackay regions.

This information has been used to inform the development of the Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009). These guidelines define trigger values that will be used to:

- Support setting targets for water quality leaving catchments;
- Prompt management actions where trigger levels are exceeded;
- Encourage strategies to minimise release of contaminants;
- Identify further research into impacts of contaminants in the Marine Park; and
- Assess cumulative impacts on the Great Barrier Reef ecosystems at local and regional levels.

It is important to note that the levels of contaminants identified in these guidelines are not targets. Instead they are guideline trigger values that, if exceeded, identify the need for management responses.

Two independent approaches were combined to define guideline trigger values for water quality:

1) Modelled relationships between the condition of reef biota, and the parameter. Secchi depth and water column chlorophyll concentration were used to identify the highest mean annual chlorophyll and lowest Secchi values that prevented high macroalgal cover and low coral and octocoral richness; and
2) Analyses of the spatial distribution of water quality in Cape York waters. Since Cape York is subject to only minor modification of land use its water quality condition was taken to be consistent with reference sites (European Community, 2005; EPA, 2006).

It is acknowledged that there are still many uncertainties in the development and application of these guidelines and, in particular, that further work is required to consider what might be achievable targets for ecosystem health given the current state of the system and level of technology. In particular, first evidence is emerging that the existence of synergistic effects may have to be carefully considered in estimates of tolerance thresholds (and hence water quality targets). For example, sedimentation effects on crustose coralline algae are significantly worsened when trace concentrations of herbicides occur in the sediments (Harrington *et al.*, 2005). Other studies have demonstrated that sedimentation effects on corals worsen with increasing organic enrichment of the sediments (Weber *et al.*, 2006), and with enrichment with marine snow (Fabricius *et al.*, 2003; Wolanski *et al.*, 2003). Studies also show that DIN enrichment enhances bleaching susceptibility (Wooldridge and Done, 2009;

Wooldridge, 2009), which is recently supported by similar findings in Florida Keys (Wagner *et al.*, 2010). It is also known that DIN enrichment exacerbates the impact of increasing ocean acidification on coral growth (Renegar and Riegl, 2005).

Seagrasses and ecosystem health

In the GBR system, seagrasses are at risk from a wide diversity of impacts, in particular where coastal developments occur (Grech, 2010; Grech *et al.*, 2010). Healthy seagrass meadows in the GBR act as important resources as the primary food for dugong, green turtles, numerous commercially important fish species and as habitat for large number of invertebrates, fish and algal species (Carruthers *et al.*, 2002). The requirements for formation of healthy seagrass meadows are relatively clear as they are photosynthetic plants occupying a marine habitat (Collier and Waycott, 2009). They require adequate light, nutrients, carbon dioxide, suitable substrate for anchoring along with tolerable salinity, temperature and pH (Collier and Waycott, 2009; Waycott and McKenzie, 2010). A number of thresholds of these requirements have been established for seagrass communities that are relevant to the GBR, and are summarised below. In addition, critical differences between responses of seagrass to changing conditions have been recognised in GBR ecosystems (Collier and Waycott, 2009) compared with those from better studied temperate ecosystems, e.g. Port Phillip Bay, Victoria (Bearlin *et al.,* 1999) and Oyster Harbour, Western Australia (Walker and McComb, 1992).

Elevated tissue nutrient concentrations in the leaves of seagrasses are indicators of excessive nutrient loads (Dennison *et al.*, 1993). The ratio of the major nutrients in seagrass tissues are indicative of the status of plant utilisation of available nutrients – when in excess, the plants are saturated and a tendency for the ecosystem to have excessive algal growth occurs (summarised in Waycott and McKenzie 2010).

N:P ratios: The ratio of N:P is also a useful indicator as it is a reflection of the 'Redfield' ratios (Redfield *et al.*, 1963), and seagrass with an atomic N:P ratio of 25 to 30 can be determined to be 'replete' (Atkinson and Smith, 1983; Fourqurean *et al.*, 1997; Fourqurean and Cai, 2001). N:P values in excess of 30 suggest P-limitation.

C:N ratios: Changing C:N ratios have been found in a number of experiments and field surveys to be related to light levels (Abal *et al.*, 1994; Grice *et al.*, 1996; Cabaço *et al.*, 2007; Collier *et al.*, 2009). Experiments on seagrasses in Queensland have suggested that at an atomic C:N ratio of less than 20, seagrass may suggest reduced light availability (Abal *et al.*, 1994; Grice *et al.*, 1996).

C:P ratios: The median seagrass tissue ratio of C:P is approximately 500 (Atkinson and Smith, 1983), therefore deviation from this value is also likely to be indicative of some level of nutrient enriched (lower C:P) or nutrient limited conditions (higher C:P).

Other indicators are more variable and reflect the interaction between local conditions and plant responses. As a result, universal threshold values have not been established; however, a matrix of responses expected under differing conditions has been developed. Further evaluation of the best indicators of seagrass health and water quality conditions has been undertaken by Waycott and McKenzie (2010) and is summarised in Section 4.5.3.

4.5.2 Indicators for monitoring and evaluation

Between 1996 and 2000, scientific staff within the GBRMPA and a number of other scientific institutions carried out significant research and monitoring programs connected with the investigation of: spatial variability of flood plume pollutant concentrations within the GBR (Devlin *et al.*, 2001); inshore GBR coral recruitment dynamics and water quality (Smith *et al.*, 2005); and temporal and spatial variation in GBR chlorophyll *a* (nutrient) concentrations (Brodie *et al.*, 2007b). More recently, the Reef Rescue Marine Monitoring Program (herein referred to as the MMP) has been implemented in the GBR lagoon to assess the long-term effectiveness of the Reef Plan, including the Reef Rescue initiative. These programs are summarised in Table 5.

Table 5. Overview of marine water quality monitoring programs in the Great Barrier Reef.

Institution	Timing	Monitoring program
GBRMPA / JCU	1994 – present	Spatial variability of flood plume pollutant concentrations within the GBR (Brodie *et al.*, 2010; Rhode *et al.*, 2006, 2008; Brodie *et al.*, 2004; Devlin and Brodie, 2005; Devlin *et al.*, 2001). Now part of the MMP (see Devlin *et al.*, 2009).
GBRMPA	1998-2001	Inshore GBR coral recruitment dynamics and water quality (Smith *et al.*, 2005)
GBRMPA / AIMS / Tourist operators	1992 – present	Long Term Chlorophyll Monitoring Program: Temporal and spatial variation in GBR chlorophyll *a* (nutrient) concentrations (De'ath and Fabricius, 2008, 2010; Brodie *et al.*, 2005, 2007b; Furnas *et al.*, 2005). Monthly collection of chlorophyll data across the GBR. Now part of the MMP (reduced coverage).
GBRMPA	1999	Spatial variability in pesticide concentrations within the GBR (Haynes *et al.*, 2000a)
GBRMPA	1999-2000	Pollutant bioaccumulation in GBR dugong and marine mammals (Haynes *et al.*, 2005; McLachlan *et al.*, 2001; Vetter *et al.*, 2001).
GBRMPA	1999	Herbicide impacts on local seagrass species (Haynes *et al.*, 2000b)
AIMS	1989-present	AIMS long-term water quality transect: Collection of water quality data at eleven coastal locations in the GBR lagoon between Cape Tribulation and Cairns (De'ath and Fabricius, 2008, 2010; De'ath, 2005; Furnas and Brodie, 1996).
AIMS	2004-present	Inshore water quality monitoring of dissolved and particulate nutrients and chlorophyll as part of the MMP using manual sampling and automated loggers at selected inshore reef sites (Schaffelke *et al.*, 2009).
UQ (EnTox)	2005-present	Pesticide monitoring using passive sampler techniques at inshore locations in the GBR (Paxman *et al.*, 2009).
CSIRO/ JCU	2005-present	Development of techniques to enable continuous monitoring of chlorophyll, suspended sediment and colour dissolved organic matter (CDOM) throughout the GBR using remote sensing techniques (also part of the MMP) (Brando *et al.*, 2010).
DEEDI / RRRC	2005-present	Inter-tidal seagrass monitoring as part of the MMP supported by the Seagrass Watch network (www.seagrasswatch.org) at estuarine, coastal and reef locations across the GBR (McKenzie and Unsworth, 2009; Waycott and McKenzie, 2010).
AIMS / Ayling and Ayling	2004-present	Inshore coral reef monitoring at 24 locations as part of the MMP (Schaffelke *et al.*, 2009; Thompson *et al.*, 2010; Sweatman *et al.*, 2007).

The MMP assesses the condition of water quality in the inshore GBR lagoon and the health of key GBR marine ecosystems – inshore coral reefs and intertidal coastal and nearshore seagrass meadows – and has been operating since 2005. The MMP has two core programs described in more detail below: (i) inshore GBR water quality monitoring, and (ii) inshore GBR biological monitoring of seagrass meadows and coral reefs, including biological indicators. In the last two years, specific event monitoring has been formally incorporated to the program to quantify peak loads of materials entering the GBR and improve understanding of material processing and ecosystem response during these important events. This information can also be used to identify changes in land runoff characteristics as a result of improvements in management practices.

During the four years of operation of the MMP, the selection of suitable water quality and ecosystem indicators has been reviewed on a regular basis, supported by a considerable amount of research that strives to enhance and streamline the design with the adoption of new indicators, largely undertaken as part of the MTSRF. In addition, technology in marine monitoring capability has improved since the commencement of the program and new, more efficient, techniques have been incorporated to the program design including pesticide passive samplers, remote sensing of indicators of nutrient and sediment concentrations, and in situ continuous data loggers to measure a range of water quality variables.

Assessing the biological health of reef ecosystems generally requires long-term assessment to confidently quantify and attribute change. In many locations of the world, long-term commitments to marine monitoring programs are recognised as fundamental to effective marine management. However, monitoring of some key marine indicators provides information on changes in water quality as a result of altered land management practices in relatively short timeframes, for example, pesticide concentrations in inshore waters and the peak concentrations of contaminants in flood events. Other longer-term indicators require measurement over extended timeframes to identify changes or trends to changing land management, such as chlorophyll and nutrient concentrations, and sediment.

The following section provides an overview of the indicators that have been established for assessing the influence of water quality on GBR ecosystems and summarises the findings of recent studies of the most suitable marine and estuarine indicators for water quality and ecosystem health in the GBR that have led to continuous improvement of the design and implementation of the MMP.

Water quality – ambient and flood events

A number of techniques are well established for monitoring marine water quality in the GBR. These range from manual grab sampling of ambient and event conditions, automated data loggers and large-scale remote sensing techniques. All of these techniques are currently employed in the MMP to describe spatial and temporal patterns in concentrations of GBR marine water quality including chlorophyll *a*, dissolved and particulate nutrient and carbon species, suspended sediment concentrations, salinity, Secchi disk values and pesticides in inshore areas. Monitoring is currently conducted at fourteen inshore sites associated with marine biological monitoring that allows for comparisons of these water quality and biological data sets.

Current water quality monitoring techniques for a range of indicators are described below (refer also to Johnson *et al.*, 2010).

Ambient water quality

Monitoring of GBR water quality in ambient conditions includes:

- **Manual water quality sampling** (by the AIMS) at fourteen core inshore coral reef monitoring sites during the wet and dry seasons for dissolved nutrients and carbon (NH_4, NO_2, NO_3, PO_4, $Si(OH)_4$), DON, DOP, DOC), particulate nutrients and carbon (PN, PP, POC), suspended solids (SS), turbidity (secchi depth), salinity and plant pigments (chlorophyll *a* and phaeophytin) (see Schaffelke *et al.*, 2009). Sampling of the six open water stations of the 'Cairns Coastal Transect', which has been undertaken by the AIMS since 1989, has also been continued.

- The monitoring of chlorophyll concentrations is still the most robust and broadly applied indicator for water quality (especially nutrient availability) in the GBR lagoon. **Inshore chlorophyll concentrations** have also been collected on a monthly basis (in most cases) using a network of community volunteers at stations along inshore-offshore transects (see Schaffelke *et al.*, 2009). Where possible, a surface water sample is collected at each site every month. Additional parameters measured at each site at the time of sampling include: salinity (with a refractometer), water temperature (with a manual thermometer), the presence of Trichodesmium blue green algae, information about the weather, wind and tides, and secchi depth and water depth (depth sounder) and the actual geographic position using a GPS.

- Placement of **autonomous instruments** (by the AIMS) for high-frequency measurements of local water quality (chlorophyll, turbidity, temperature and light) at all 14 water quality and inshore reef sites (see Schaffelke *et al.*, 2009). The Eco FLNTUSB combination instruments perform simultaneous *in situ* measurements of chlorophyll fluorescence, turbidity and temperature at ten minute intervals. Data time series were obtained for all fourteen deployment locations, with some data gaps. Time-series data are summarised as daily means, calculated from the readings obtained every ten minutes. Instrumental data are validated by comparison with chlorophyll and suspended solid concentrations obtained by analyses of water samples collected close to the instruments, carried out at each change-over. Turbidity is measured simultaneously by detecting the scattered light from a red (700 nm) light-emitting diode at 140 degrees to the same detector used for fluorescence. The instruments were used in 'logging' mode and recorded a data point every ten minutes for each of the three parameters, which was a mean of fifty instantaneous readings.

- The broad application of **remote sensing techniques** (by the CSIRO and JCU) has been investigated through the MMP and is showing large potential to be a cost-effective method to determine spatial and temporal variation in near-surface concentrations of suspended solids (as non-algal particulate matter), turbidity (as vertical attenuation of light coefficients Kd), chlorophyll *a* and coloured dissolved organic matter (CDOM) for the GBR (see Brando *et al.*, 2010). This is achieved through the acquisition, processing (with regionally valid algorithms), validation and transmission of geo-corrected ocean colour imagery and data sets derived from MODIS imagery. Further support to continue and improve the availability of validation data for remote sensing results, particularly in offshore and Far Northern areas of the GBR, and better understanding of the limitations with remote sensing applications in the wet season due to frequent cloud cover will assist in progressing this technology as a key component of the MMP.

- Time integrated baseline concentrations of specific organic chemicals in water are collected with the aim to evaluate long-term trends in **pesticide concentrations** (by the National Research Centre for Environmental Toxicology, University of Queensland) in inshore waters of the GBR, using passive sampler techniques. These techniques have been developed through the MMP (see Paxman *et al.*, 2009), and are briefly explained here. Grazing and cropping (in particular sugar cane) account for a significant proportion of land use in the GBR catchments (Brodie and Waterhouse, 2009). Pesticides commonly

used in these industries include organophosphates (e.g. chlorpyrifos) and triazines (e.g. atrazine, simazine, ametryn, prometryn) as well as urea-based herbicides (e.g. diuron, tebuthiuron, flumeturon) (Lewis *et al.*, 2009b). Depending on the physical properties of these pesticides, their mobility and half lives vary, but those that are persistent and mobile have the potential to be transported from the sites of application in the catchment via rivers into the marine environment. Many of these pesticides occur at trace levels that are very difficult to detect and quantify, yet these low concentrations may ultimately pose a chronic risk to the environments they contaminate. Time integrated passive sampling techniques have been developed for the monitoring of trace organic pollutants in water (Shaw and Mueller, 2005; Shaw *et al.*, 2010; Paxman *et al.*, 2009). When deployed for an extended period of time (30-60 days) these samplers can accurately predict average water column concentrations of a range of pesticides. This is the primary approach adopted for pesticide monitoring in the MMP.

Pesticide concentrations are measured at thirteen inshore reef sites using passive samplers. These samplers are deployed for approximately thirty days during the wet season (November to March) and for two month periods during the dry season (April to October). In addition, as part of a new toxicology investigation, samplers were deployed for the collection and concentration of pesticides for toxicological testing at twelve inshore reef sites during the coral spawning season. This component of the MMP was originally designed to collect baseline data on pesticides in the GBR in terms of presence and extent, and is gradually progressing to improve understanding of the spatial and temporal distribution of pesticides in the GBR. The data also contribute to an improved understanding of the ecological effects of pesticides on GBR ecosystems.

Flood plume water quality

Event monitoring of flood plume waters provides an assessment of the distribution of concentrations and major land-sourced pollutants in the GBR lagoon during flow events and quantifies the exposure of GBR ecosystems to these contaminants. However, due to the large size of the GBR, the short-term nature and variability (hours to weeks) of runoff events and the often difficult weather conditions associated with floods, it is very difficult and expensive to launch and coordinate comprehensive runoff plume water quality sampling campaigns across large sections of the GBR. To counter this variability, the MMP has adopted a multi-pronged approach in the assessment of the exposure of the GBR inshore coral reefs to materials transported into the lagoon from GBR catchment rivers, represented in Figure 15 (see Devlin *et al.*, 2009).

Current indicators and techniques for monitoring of GBR water quality in event conditions (see Devlin *et al.*, 2009) include:

- **Manual water quality sampling** (by JCU) undertaken in the flood plume waters. Depth profiles using a Hydrolab are collected at most locations for pH, salinity, dissolved oxygen and turbidity, and a new chlorophyll probe has been trialled, although equipment failures resulted in data gaps on some sampling campaigns. Surface water samples are collected at all sites for dissolved nutrients (NH4, NO2, NO3, PO4, DON, DOP), particulate nutrients (PN, PP), suspended solids (SS), plant pigments (chlorophyll a and phaeophytin) and CDOM. Samples are also collected at selected sites for pesticides, phytoplankton counts, trace metals and sediment characteristics.
- Remote sensing techniques are applied (by JCU and the CSIRO) to assist in understanding the movement, extent and duration of flood plumes. In 2008/09, true colour images were extracted to identify the extent of the riverine plume, available algorithms were applied to satellite images to extrapolate chlorophyll and colour dissolved organic matter (CDOM) data for the appropriate images, and imagery was used as a

near-real time tool to guide field sampling, with imagery processed on a daily basis to provide information of plume movement to scientists taking in situ samples.

• Extent and exposure of plume waters (by JCU and CSIRO) is estimated using aerial images from 1994 to 1999 combined with remote sensing images from 2002 to 2009 to describe the full extent of riverine plumes. This has been completed for the Tully River during eleven events and the Burdekin River during seven events. The derived CDOM absorption at 412 nm combined with careful examination of quasi-true colour and chlorophyll a images provides the information used to define river plume 'type' (primary, secondary and tertiary) and extent. Plume exposure mapping is then produced using a combination of plume classification and ArcMap geoprocessing. The results of this work are described in more detail in the reports, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.) and 'Priority Pollutants in the GBR' (Waterhouse and Brodie, in prep.).
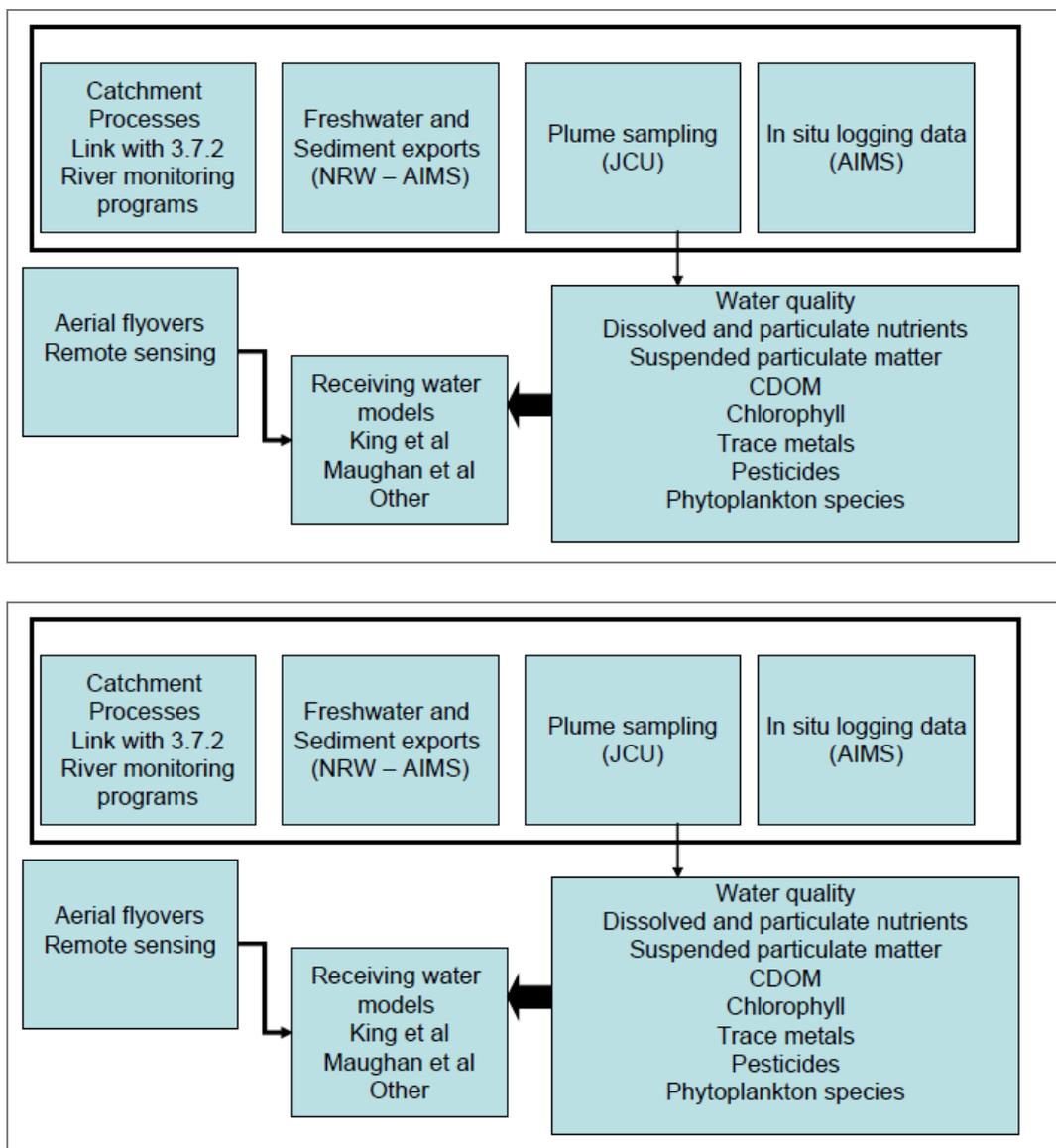
Figure 15. Diagrammatic representation of the integrative programs running concurrently with the marine flood plume monitoring program. Source: Devlin *et al.* (2009).

Estuarine health

Presently, estuarine health is not measured as part of any strategic monitoring and evaluation program for the GBR catchment, although some water quality measurements (dissolved oxygen, temperature, pH, conductivity, turbidity, chlorophyll *a*, nitrogen and phosphorus – total and dissolved) are collected monthly in the Fitzroy and Burnett estuarine areas as part of the Paddock to Reef Program.

Sheaves and others (2007) assessed techniques that can be employed to determine the ecosystem health of estuaries and coastal wetlands in Australia's tropical regions, evaluated the sensitivity of those techniques to detect the effects of specific stressors, and evaluated their ability to separate natural variations from deleterious anthropogenic impacts. The study showed that while there is a large amount of information about detecting impacts and measuring ecosystem health in temperate estuaries, the extent to which temperate approaches are transferable to tropical/subtropical systems is unclear. There have been no location-specific studies evaluating the appropriateness of extrapolation from temperate to tropical understanding. In particular, biochemical processes such as toxicity, persistence and accumulation rates are likely to differ between cooler temperate and warmer tropical systems. Contrasts in functioning of tropical compared to temperate estuaries are likely to be compounded by the much higher biological diversity present in tropical estuaries, which potentially leads to more complex ecological processes High diversity might also equate to high variability, adding another layer of complexity.

Investigations undertaken through the MTSRF have aimed to fill some of these knowledge gaps. Key findings outlined below (Section 4.5.3) on the development of new and innovative indicators for the GBR, are relevant to the GBR and other tropical ecosystems.

Marine ecosystem health

As indicated above, extensive research has shown that land-based water quality pollutants can have potentially deleteriously impacts on sensitive marine ecosystems that are found in the inshore areas of the GBR, such as coral reefs and seagrass meadows (e.g. Fabricius, in press, 2010; Fabricius *et al.*, in prep.; Waycott and McKenzie, 2010; De'ath, 2007; Negri *et al.*, 2005; Fabricius, 2005; Haynes *et al.*, 2000a, 2000b). Monitoring of these marine ecosystems that are recognised as being most at risk from land-based pollutants is undertaken as part of the MMP to assess their current condition and to identify any trends in their status over time.

Seagrass health

The inshore seagrass monitoring program quantifies temporal and spatial variation in the distribution of intertidal seagrass meadows and correlates, where possible, seagrass status with change in delivery of land-sourced contaminants. The core part of the program is conducted through a community volunteer program known as Seagrass–Watch as well as additional scientist collected parameters. This approach enables a more broad-scale monitoring of the GBR than would otherwise be possible with available funding. Seagrass monitoring sites have been located as close as practicably possible to river mouth and inshore marine water quality programs (dependent in some cases on historical monitoring and location of persistent seagrass meadows) to enable correlation with concurrently collected water quality information.

The following indicators are currently incorporated into the MMP (by DEEDI and JCU):

- The **status of intertidal seagrass meadows** is monitored bi-annually at thirty sites in fifteen locations (estuary, coastal and reef locations) between Cooktown (until 2009) and Hervey Bay (see McKenzie and Unsworth, 2009). Sites are monitored for seagrass percent cover, species composition and meadow area (edge mapping of the immediate

area). Additional information is collected at each site for canopy height, within-canopy temperature, algae cover, epiphyte cover and macrofaunal abundance.

- To assist in correlation of seagrass status with change in delivery of land-sourced material, supporting *in situ* **water quality** information, including seagrass tissue nutrients, sediment nutrients and sediment herbicides (until 2009) was collected at all locations (see McKenzie and Unsworth, 2009). Seagrass canopy light was also measured at inshore and offshore locations in the Cairns and Townsville locations.

- **Seagrass resilience** is being measured as the ability for seagrass habitats to recover following disturbances and is linked essential given the high disturbance levels they are exposed to. Resilience is linked to their ability to produce seeds and therefore reproductive effort is being used as an indicator of the resilience of seagrass meadows. Two measures of seagrass reproduction are recorded at each site: the presence of seeds, and live plant reproductive effort (the number of reproductive structures – spathes, fruit, female flower or male flowers – per seagrass node) (see Waycott, 2010).

Coral health

The reef monitoring sites are close to the sampling locations for lagoon water quality to assess the relationship between reef communities and water quality as well as other, more acute impacts. Within each region, reefs are selected that represent a gradient in exposure to runoff, largely determined as increasing distance from river mouth in a northerly direction. To account for spatial heterogeneity of benthic communities within reefs, two sites are selected and stratified by depth. Within each site and depth fine-scale spatial variability is accounted for by the use of five replicates. Reefs within each region are designated as either core or cycle reefs. Core reef locations have annual coral reef benthos surveys, coral settlement assessments, autonomous water quality instruments (temperature, chlorophyll and turbidity) and regular water quality sampling. Non-core (cycle) reef locations have benthos surveys every two years, and no water quality assessments. Exceptions are Snapper Island (water quality instruments, regular water sampling, coral annual surveys, but no coral settlement) and Dunk Island (water quality instruments, regular water sampling, but coral surveys every other year).

The following indicators are currently incorporated into the MMP (by the AIMS):

- The **status of inshore coral reefs** is assessed at 24 inshore reef locations in four NRM regions: the Wet Tropics, Burdekin, Mackay-Whitsunday and Fitzroy regions. The coral monitoring surveys the cover of benthic organisms, the numbers of genera, the number of juvenile-sized coral colonies and sediment quality at each location (e.g. see Schaffelke *et al.*, 2009; Thompson *et al.*, 2010).

- **In situ water quality** sampling is routinely carried out at all reef monitoring sites to allow correlation with reef condition (see Schaffelke *et al.*, 2009).

- **Coral reef resilience** is measured using coral recruitment as an indicator combined with the above information collected on current and past status. Coral recruitment monitoring is undertaken at three core sites in each of the four NRM regions using settlement plates (see Thompson *et al.*, 2010).

- Assessments of **sediment quality** and assemblage composition of **benthic foraminifera** (a water quality bioindicator in the testing phase; see Section 4.5.3) are new components of the coral reef monitoring which provided additional information about the environmental conditions at individual reefs.

- **Pesticide phytotoxicity** testing is incorporated to improve understanding of the environmental relevance of the presence of pesticides at inshore reefs but also to better integrate the pesticide data with biological monitoring data. This component is based on the use of passive sampling techniques co-located at coral recruitment sites during the spawning season. Samplers are analysed by assessing the inhibition of photosynthesis in

algae (ideally isolated zooxanthellae) that are dosed with concentrated extracts from passive samplers that were exposed at selected sites over the coral spawning season. The technique involves deployment of 'double disk' polar passive samplers in a standard housing at twelve coral reef monitoring sites for periods ranging from 52-67 days (October to December) (see Schaffelke *et al.*, 2009).

4.5.3 Testing new and innovative techniques

A majority of new work related to investigating, testing and refining marine water quality indicators is focused on indicators of ecosystem health. An overview of the investigation of optimal ecosystem health indicators through the MTSRF is provided below. Understanding material generation, delivery and fate is also important for understanding ecosystem response, and hence, status. This report provides a brief overview of activities related to this task and a more comprehensive review is provided in the companion report, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.). Innovative techniques are also being investigated for broad-scale water quality measurements, particularly in the area of pesticide monitoring, phytoplankton communities and remote sensing applications through the MMP (e.g. Brando *et al.*, 2010; Devlin *et al.*, 2009)*.*

Indicators of material delivery and fate

Understanding material generation, delivery and fate has been a substantial focus of research activities in the former Catchment to Reef Joint Research Program as well as the MTSRF. The following section provides a brief overview of the MTSRF research on coral cores, and flood plume exposure assessments undertaken through the MMP. Additional work is reviewed in the companion report, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.).

Coral cores

Coral core records provide excellent insights into changing water quality in the GBR lagoon since European settlement (Jupiter, 2006; Jupiter *et al.*, 2007, 2008; Lewis *et al.*, 2007c; 2010; Marion, 2007; McCulloch *et al.*, 2003a, 2003b). The setting of targets for suspended sediment, nutrient and pesticide runoff in the GBR catchments requires prior knowledge of current and baseline (pre-European settlement) exports. Coral records provide a means of assessing sediment and nutrient runoff over long timeframes to assist in this process (Lewis *et al.*, 2010). Recently, coral proxies (Ba, Y, Mn) have provided evidence of increased sediment export to the GBR lagoon from the Burdekin River catchment (Lewis *et al.*, 2007c; McCulloch *et al.*, 2003a, 2003b) while nitrogen isotopes in the (insoluble) organic component of the coral skeleton have been used to quantify increases in nutrient loads from the Pioneer River (Jupiter *et al.*, 2007, 2008; Marion, 2007). Changes in the nitrogen isotopic signature and coral nitrogen concentrations were correlated with increased fertiliser application in the Pioneer River catchment (Jupiter *et al.*, 2007, 2008; Marion, 2007), although additional studies from other parts of the GBR as well as an understanding of N isotope dynamics in river water plumes are required to validate these findings.

More recent MTSRF research in the Whitsunday Islands (see Lewis *et al.*, 2010) and Dunk Island (see Mallella *et al.*, 2010) confirms that coral cores are a useful tool for indicating long-term changes to water quality at specific locations which can be linked to river discharge characteristics. This tool could therefore form a useful component of an overall monitoring and evaluation toolkit for the GBR, particularly to inform target setting for pollutant loads.

Flood plume extent and exposure

The extent of river plumes in the GBR (along the GBR and cross-shelf) is a consequence of several factors, including river flow (volume and duration), wind direction and velocity,

currents and tidal dynamics. The extent and duration of flood plumes can have significant implications for the health of inshore marine ecosystems, such as seagrasses and coral reefs. The dynamics of a flood plume as it moves from the river mouth into the marine environment can be described in terms of the hydrological and chemical behaviour. At first flood plumes contain elevated concentrations of sediments (and associated nutrients and pesticides). Later, when particulate matter falls out of the plume waters the plume is characterised mainly by presence of the dissolved materials (and the associated nutrients) (Brando *et al.*, 2010).

In flood plumes, coloured dissolved organic matter (CDOM) concentrations (measured through remote sensing techniques) are high and largely derived from terrestrial sources, making CDOM a useful tracer of terrestrial discharge of low salinity waters (Brando *et al.*, 2010). The flood extent can be estimated by applying a threshold to the maps of CDOM seasonal maximum values. The extent is defined by applying a threshold of CDOM of 0.2 nm^{-1} derived by performing a qualitative analysis of the relationship between in-situ CDOM absorption and salinity, excluding data from around reefs which would include marine-derived CDOM. This extent represents the maximum influence of fresh water due to the strong relationship between CDOM and the adsorption curve, with a distinct marine signal.

Maps of the frequency of plume exposure and estimates of associated risks have also been developed through MTSRF research and the MMP (see Devlin and Schaffelke, 2009; Devlin *et al.*, 2010). The assessments categorise the plumes into zones of relative risk (low to high) and estimate the number of ecosystems exposed within those zones. Further work is required to incorporate improved hydrodynamic modelling into these assessments to provide a better estimation of material transport and, hence, fate.

Marine ecosystem health

Coral bioindicators

The use of bioindicators in ecosystem health monitoring programs can provide advantages over direct measurements of water quality. Bioindicators provide a time-integrated measure (from time periods of minutes to years) of the effects of changes in water quality on coral reefs. This is useful if water quality sampling is discontinuous and weather-dependent, so that episodic events that can strongly influence the structure of coral communities may be missed (e.g. floods or sediment re-suspension during strong winds). Combining a suite of bioindicators of cellular, organism and community effects will more effectively attribute ecological change to changes in specific environmental conditions than the use of a single indicator (Erdmann and Caldwell, 1997; Jameson *et al.*, 1998; Cooper *et al.*, 2009).

In the early stages of the MTSRF, coral-based indicators at a range of spatial and temporal scales were quantitatively assessed and compared, and those most suitable for inclusion into a 'toolbox' for monitoring the health of nearshore reefs on the GBR were identified (Cooper and Fabricius, 2007). This work was then extended more broadly, to review the suitability of a range of bioindicators for use in monitoring programs that link changes in water quality to changes in the condition of coral-reef ecosystems (Cooper *et al.*, 2009). From the literature, 21 candidate bioindicators were identified, whose responses to changes in water quality varied spatially and temporally. Responses ranged from rapid (hours) changes within individual corals to long-term (years) changes in community composition and are summarised in Cooper *et al.* (2009). From this list, the most suitable bioindicators were identified by determining whether responses were (i) specific, (ii) monotonic, (iii) variable, (iv) practical and (v) ecologically relevant to management goals. These criteria, which could be applied in the selection of indicators for any ecosystem health monitoring program, are described in Table 6.

Table 6. Criteria for the selection of bioindicators to assess effects of changes in water quality on corals and coral communities. Source: Cooper *et al.*, 2009. Modified from Jones and Kaly, 1996; Erdmann and Caldwell, 1997; and Jameson *et al.*, 1998.

Criteria	Definition
Specificity	Biological response is specific to the stressor of interest and not to other environmental stressors
Monotonicity	The magnitude of the biological response should reflect the intensity and duration of the stressor of interest
Variability	Biological responses should be consistent at a range of spatial and temporal scales. Ideally, there should be low background variability although a change in variance can itself be used as an indicator of an impact
Practicality	Measurements of biological responses should be cost effective, easy to measure, non-destructive and observer independent
Relevance	Biological response should be ecologically relevant and important in public perception to assist communication

For long-term monitoring programs that aim to quantify the effects of chronic changes in water quality, eleven bioindicators were selected: symbiont photophysiology, colony brightness, tissue thickness and surface rugosity of massive corals, skeletal elemental and isotopic composition, abundance of macro-bioeroders, micro- and meiobenthic organisms such as foraminifera, coral recruitment, macroalgal cover, taxonomic richness of corals and the maximal depth of coral reef development. For short-term monitoring programs, or environmental impact assessments that aim to quantify the effects of acute changes in water quality, a subset of eight of these bioindicators were selected, including partial mortality. Their choice will depend on the specific objectives and the timeframe available for each monitoring program.

Each of these measures has a different sensitivity and response time to changes in water quality. A combination of these measures, complemented by indicators based on biofilms and direct water quality measurements, is therefore recommended as a composite indicator system to assess changes in the exposure and condition of nearshore reefs on the GBR. The indicator measures were plotted against increasing levels of stressors in ascending order, from sub-lethal stress to mortality (Figure 16), providing an assessment framework to assist in the selection of bioindicators to quantify the effects of changing water quality on coral-reef ecosystems. These responses vary according to the magnitude and duration of exposure to the stressors. Similar responses to those presented in the short-term may occur following exposure to lower levels of stress over longer (months to years) periods of time but this remains to be determined. Exposure to the key components contributing to decreased water quality (i.e. elevated sediments, turbidity and nutrients, and reduced irradiance) will first invoke a response at the physiological level (i.e. the early warning indicators). At increasing exposure (either longer duration or higher levels), responses at the population and community level will become evident.

Testing of these indicators in the GBR has been undertaken through the MTSRF, supported by the MMP, through two independent studies. The first study was conducted along one environmental gradient in the Whitsundays (Study 1), while the second study was conducted on twelve reefs located at increasing distance from rivers in four GBR regions (at MMP sites) (Study 2) (refer to Fabricius *et al.*, 2010a for further detail). The trials demonstrated that some coral reef indicators consistently change along water quality gradients in four regions of the GBR, and confirm that some of the indicators previously proposed for the Whitsundays (Fabricius *et al.*, 2007; Cooper *et al.*, 2007) are indeed valid across the GBR.

Ecosystem response		Components of water quality		
Response and recovery time	Level of response	Nutrients	Sedimentation	Turbidity and Light attenuation
Minutes – days	Colony	**Symbiont photophysiology**		
		Increased photosynthesis	Decreased maximum quantum yield	Decreased minimum saturating irradiance, increased light utilisation coefficient
		Coral brightness		
		Darkening	Paling	Darkening
		Tissue thickness of massive *Porites*		
		Increased	Decreased	Decreased
		Surface rugosity of massive *Porites*		
		Increased		
		Skeletal elemental and isotopic composition		
		Elevated δ^{15}N (sewage effluent); low δ^{15}N (fertilisers)		
Weeks – months	Population	**Macro-bioeroders**		
		Increased		Increased
Months – years	Community	**Micro- Meiobenthic bioindicators (FORAM Index)**		
		Decreased large symbiont-bearing foraminifera		Decreased large symbiont-bearing foraminifera
		Coral recruitment		
		Reduced	Reduced	Reduced
		Benthic cover		
		Increased cover of macroalgae		Increased cover of macroalgae
		Taxonomic richness		
		Reduced	Reduced	Reduced
		Maximum depth of coral-reef development		
				Reduced

Increasing stress and degradation of water quality

Figure 16. Conceptual model of coral bioindicators to indicate increasing exposure to the key components of water quality. Responses are presented in increasing order of effect from stress to mortality resulting from increasing levels of stressors. Responses will depend on both the magnitude and duration of changes in the levels of stressors (e.g. Kuntz *et al.*, 2005). All the responses will first be evident at the genetic/colony level and then in the wider community. Sublethal responses, therefore, may pre-empt more severe effects at the population and community level and can be used to describe shifts in ecosystem condition from healthy (green) to degraded (red) conditions. Source: Cooper *et al.* (2009).

Ten measures were consistently and strongly related to water quality in both studies:

- In living *Porites*: colony brightness and externally visible macrobioeroder densities;
- In *P. damicornis*: chlorophyll *a* content per unit branch surface area;
- In foraminifera: the Foram Index;
- In benthic indicators (strongly declined with increasing values of chlorophyll and turbidity):
    - three measures of macroalgal cover: total cover, and the cover of brown and red algae;
    - *Acropora* cover;
    - the ratio of *Acropora* to total hard coral; and
    - *Turbinaria* cover.
- In coral juveniles: soft coral juvenile density. Hard coral juvenile density and diversity were also strongly related to water quality in Study 1, but only weakly declined in Study 2 where regional effects were dominant. The latter deserves closer inspection with potential covariates unrelated to water quality such as; the availability of suitable substratum not already occupied by other benthos, and regional variation in broodstock.

In contrast, partial mortality of massive *Porites* did not differ in response to water quality, which contrasts with some previous findings (Nugues and Roberts, 2003). However, this is not surprising as various other environmental factors can cause partial mortality, including bleaching, coral disease or crown-of-thorns starfish predation.

Some of the recommended measures are not as easily obtained as others. In particular, the determination of chlorophyll *a* in *P. damicornis* requires destructive sampling and time consuming laboratory analyses. Since *Porites* brightness may be considered as a proxy for chlorophyll content and data are much more easily obtained using a colour chart, the former measure is excluded from the final list of indicators. Brown and red macroalgal cover are a subset of total macroalgal cover, and since the reliability of determining phyla from photographs may vary between observers, only total macroalgal cover may be used as an indicator. The determination of juvenile densities is time consuming in the field: however due to its environmental relevance (determining the time of recovery from disturbance), all four coral juvenile measures are retained in the list of final indicators.

The final list of water quality bioindicators is therefore:

1. *Porites* brightness
2. Macro-bioeroder density in massive living *Porites*
3. Foram index
4. Macroalgal cover
5. *Acropora* cover
6. *Acropora*/hard coral cover ratio
7. *Turbinaria* cover
8. Hard coral juvenile density
9. Hard coral juvenile richness
10. Soft coral juvenile density
11. Soft coral juvenile richness

Each bioindicator has a different level of sensitivity and response time. Coral brightness changes within about twenty days to changing water quality, and is hence useful as relatively

rapid indicator of acute changes. In contrast, community measures and bioeroder densities change on a time scale of years and are hence a better indicator of long-term changes in exposure (Cooper *et al.*, 2009).

The methods used to measure these parameters are rapid and non destructive, and based on four sets of data: photo transects, the assessment of *Porites* for brightness and bioerosion, sediment samples for foraminifera, and the survey of coral and octocoral juvenile density and diversity. Three of the four data sets are already part of the routine MMP. The colour reference chart is the quickest way to monitor changes in the brightness of *Porites*. The count of bioeroders on coral surface is done very easily and quickly with a quadrat placed on the coral. The method used to measure surface rugosity via the depth of grooves will require improvements to obtain more accurate measurements. Only the assessment of photo transects and juvenile densities require a high level of expertise for taxonomic identification and a consistency in finding the small and often semi-cryptic coral juveniles in the field. To support the implementation of this indicator set, Standard Operating Procedures and field sampling protocols have also been developed and are described in Fabricius *et al.* (2010b).

The direct measurement of water quality will always remain the highest priority. However, such loggers are expensive and long-term records are presently only available from less than twenty inshore reefs of the GBR. On the hundreds of inshore reefs without turbidity logger data and for as long as remote sensing turbidity estimates are not yet operational, the proposed set of bioindicators will allow an estimate of past water quality exposure based on a rapid field sampling protocol. A final development of a robust and reliable indicator system may eventually allow assessment of changes in water quality on other inshore reefs of the GBR that are presently not instrumented with water quality loggers.

Researchers are now in the process of developing a statistical tool to use the two sets of data (water quality and biota, both with many variables and few cases) to predict water quality from a set of bioindicators, and vice versa. This is based on a combination of the data from both Studies 1 and 2. This new statistical method is presently being developed (Fabricius *et al.*, in prep.), and will represent a tool box to convert values from a set of indicators to a single turbidity value and its confidence intervals.

Biofilms

Microbial communities are potential indicators for water quality as they respond rapidly to environmental changes. Investigations of microbial biofilm communities were conducted in coastal and offshore areas of the Whitsunday Island group (Kriwy and Uthicke, in prep.; Witt *et al.*, 2010). Analyses showed distinct microbial assemblages between offshore and nearshore communities. In general, communities shift towards an increase in nitrogen-fixing Cyanobacteria and phototrophic members of the *Roseobacter clade*, under high light/low nutrient conditions (e.g. offshore, and to some extent during the dry season). *Gammaproteobacteria*, especially sulphur oxidising members, show the inverse trend, i.e. they are more abundant at inshore sites and during the wet season. Net production of biofilm microbial communities was higher in the wet season than the dry season, and higher offshore than inshore. Dominance shifts in key microbial groups in biofilm communities have the potential to eventually become useful candidate bioindicators of tropical coastal water quality.

Forams and coral assemblages

Benthic foraminifera are established indicators of marine and estuarine pollution in temperate regions (Alve, 1995) and have been applied as indicators of water quality in Florida and the Caribbean using a simple index, the 'FORAM index' (Hallock, 2000; Hallock *et al.*, 2003). Shifts in the index over time have coincided with general reef degradation caused by land

runoff (Hallock *et al.*, 2003). In recent years Uthicke and others have explored the application of the FORAM index to the GBR and the application of foraminifera as indicators of water quality in the GBR. The FORAM index corresponded well to a water quality gradient in the GBR, suggesting that decreased light, increased inorganic nutrients and organic matter availability may cause a shift towards higher contribution of heterotrophy (Uthicke and Nobes, 2008; Schueth and Frank, 2008). The studies included investigation of the distribution of benthic foraminifera in four regions of the GBR and along a water quality gradient in the Whitsunday region; and manipulative laboratory experiments to determine whether the distribution of symbiont bearing foraminifera is controlled by light levels (Uthicke and Nobes 2007; Uthicke *et al.*, 2010). Coral assemblages were also investigated.

Environmental variables (i.e. several water quality and sediment parameters) and the composition of both benthic foraminiferal and hard-coral assemblages differed significantly between four regions (Whitsunday, Burdekin, Fitzroy, and the Wet Tropics). The observed spatial patterns for foraminiferal and coral assemblages showed significant similarity. A significant amount of variation in the foraminiferal distribution was explained by sediment properties (the proportion of very fine sands and fine sands (63-250 μm grain size) and clays and silts (<63 μm), organic matter and inorganic carbon content) and by turbidity and concentrations of particulate matter in the water column. Heterotrophic species of foraminifera were dominant in sediments with high organic content and low light, whereas symbiont bearing mixotrophic species were dominant elsewhere. A similar suite of parameters explained 89% of the variation in the FORAM index and 61% in foraminiferal species richness. The FORAM index (high values = high relative abundance of symbiont-bearing taxa) decreased with increasing proportions of sediments with small grain sizes and high organic matter content and with increasing concentrations of water column particles (and hence reduced light availability). In contrast, this index increases with increasing values of sediment inorganic carbon and increasing hard coral cover. Variation in foraminiferal taxa richness was also explained by environmental parameters and mainly increased with increasing proportion of sediments with small grain sizes.

Coral assemblages varied in response to environmental variables. The proportions of sediments with small grain sizes (<63 μm and between 63-250 μm), the combined organic carbon and nitrogen content of the sediment and the composite water column parameter 'particulates' explained most of the variation in coral assemblage composition. Coral assemblages are very dynamic and constantly change due to acute disturbances such as cyclones, outbreaks of *Acanthaster planci*, coral bleaching and disease. Coral assemblages are also shaped by the chronic settings of their environment, which, for example, influence coral recruitment and impede coral growth. As a result of acute and chronic causes, very different coral communities may occur at sites with similar environmental conditions. Thus, it is likely that a mosaic of coral assemblages with different acute disturbance histories obscures the perhaps more subtle effects of environmental quality that foraminifera detect. Some of these acute disturbances, especially temperature and light conditions leading to bleaching, might also affect foraminiferal assemblages. However, it is unlikely that effects of cyclones, for instance, are as severe. In addition, smaller size and faster turnover will allow foraminiferal assemblages to recover and reach successional endpoints more rapidly.

Coral cover was positively related to the FORAM index in the inshore reefs investigated. While this could be interpreted as additional support for the validity of the FORAM index as an indicator for reef health, further work over longer time scales is required to test whether the FORAM index would also track changes in coral cover over time. There was no significant relationship between the richness of coral genera and any of the environmental parameters.

To conclude, it is proposed that assemblage composition of foraminifera, but not of corals, is a useful indicator of short-term (years) changes in environmental quality. While coral

assemblage composition varied in different sediment and water quality conditions, we cannot fully interpret these changes until the ecology of a wide range of coral taxa is better understood. However, future research and monitoring of coral population dynamics, especially recruitment and rate of recovery after disturbances under different environmental regimes, continues to be important, because hard-coral cover and diversity are important conservation targets. Foraminifera are likely to respond faster to changes in water quality and are less susceptible to acute catastrophic disturbances. In contrast, coral data are more difficult to interpret because they are shaped by acute disturbances, and surprisingly little is known about the basic ecology of individual species. Foraminiferal assemblages are effective bioindicators of turbidity/light regimes and organic enrichment of sediments on coral reefs. Fine-tuning of the FORAM index will require further studies of foraminiferal ecology, especially with regard to host–symbiont relationships.

More advanced investigations of foraminifera have revealed potential additional applications in water quality monitoring. For example, Uthicke and others (2010) investigated whether symbiont-bearing benthic foraminifera are good model organisms to test whether exposure to land runoff increases vulnerability to climate change. The two species tested (in situ and in aquaria) showed significantly reduced growth rates on reefs closer to the mainland with higher exposure to land runoff. Aquaria experiments were also conducted manipulating both temperature and nutrient availability. At water temperatures >2°C over current average summer levels, both species had reduced photosynthetic yields and reduced chlorophyll concentration, suggesting an expulsion of symbionts similar to that observed in corals ('bleaching'). Increased temperatures significantly reduced growth in both species. Responses to increased nitrogen concentrations were significant in one specifies with reduced growth and increased mortality. Thus, foraminiferal species with different types of symbiont responded differently to increased nutrient conditions. However, the species with symbionts similar to corals showed additive stress effects of temperature and nutrients for both mortality and growth. This suggests that, at least for this species, improved local management of agricultural runoff would ameliorate potential effects of global stressors such as climate change.

Uthicke and Altenrath (in press) have also investigated the effect of water column nutrients on growth and C:N ratios of symbiont-bearing benthic foraminifera in inshore and offshore reefs. Concentrations of most dissolved and particulate water quality variables were significantly higher inshore and during periods of high runoff (summer wet season). Foraminiferal growth was generally significantly lower on inshore reefs than on offshore reefs and growth of both species was reduced during the wet season. Depth-transplantations confirmed that light was not an important factor in growth regulation. In contrast, multiple regression analyses of the effects of water quality variables on foraminiferal growth explained 69% of the variance in growth for *A. radiata*, and 78% for *H. depressa*. Increased concentrations of dissolved nitrogen were associated with reduced growth. Intracellular carbon to nitrogen ratios in the foraminifera also reflected patterns in water quality, with generally lower values in foraminifera from inshore or during periods of high runoff, driven by higher intracellular nitrogen contents during these periods. It is suggested that increased nutrient availability releases foraminiferal symbionts from nutrient-limitation. This may lead to reduced translocation of organic carbon to the host, and resulting reduced host growth.

Diatoms

The use of benthic diatoms as indicators of water quality has also been investigated through the MTRSF. Despite their ecological importance, very little is known about the taxonomy and ecology of benthic diatoms in coral-reef ecosystems (Gottschalk *et al.*, 2007). Diatom densities and community compositions were investigated in three distinct regions of the GBR: Wet Tropics, Princess Charlotte Bay, and the Outer Shelf. The analysis revealed significant differences in community composition between all three regions, with indications

that variations in light and nutrient availability are the most likely explanation for the observed differences. These results warrant further investigation of the potential application of diatoms as an indicator of water quality in the GBR.

Seagrass health

Theoretically, habitats go through a series of cascading changes as the result of external stressors (Figure 17). In seagrasses, short-term changes would be a physiological change or even a subtle change in the morphology of new plant structures. Longer-term changes or high amplitude impacts may result in gross morphological change to the plants themselves. The greatest changes occur when there are limits to individual plants actually surviving in the population, resulting in community change or even loss of a seagrass community altogether (Figure 17).

Figure 17. Cascade of seagrass responses to increasing stress, such as reduced light availability (adapted from Waycott *et al.*, 2005).
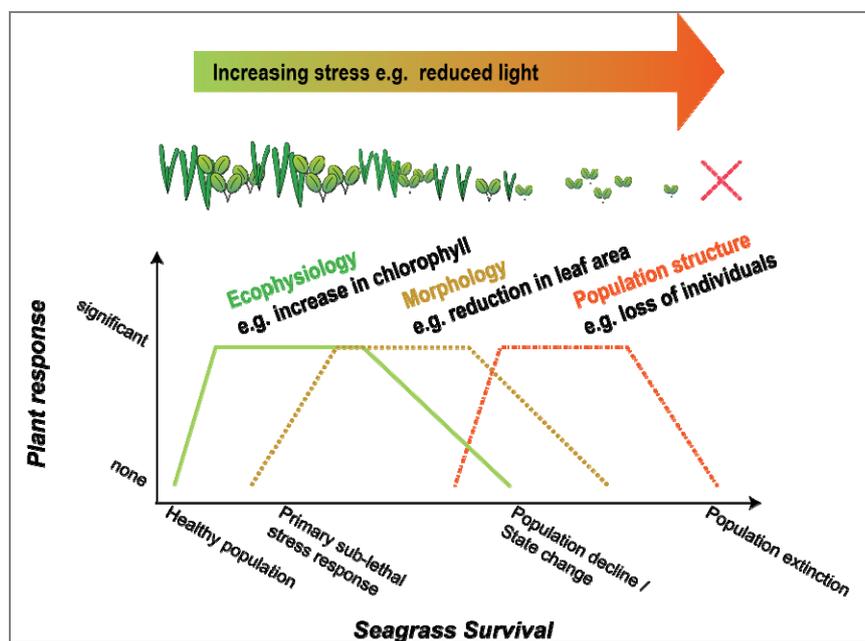
Using this model of cascading change and through understanding and observing changes at the various stages of plant and population response, the status of a seagrass meadow can act as an early warning system of impending state change (e.g. Dennison *et al.*, 1993; Orth *et al.*, 2006; Waycott *et al.*, 2009).

Using the different stages of seagrass meadow response, the potential for various measurable characteristics of seagrass meadows to act as indicators of ecosystem health has been evaluated as part of the MTSRF and the results are outlined in Waycott and McKenzie, (2010).

As seagrasses grow rooted in place and possess relatively long-lived tissues they integrate temporally variable environmental signals such as fluxes in nutrients, toxicants and light availability. Because of this, it can be more useful to sample seagrass tissues to evaluate long-term average elemental availability than it would be to sample the variable environment for the same element. Among GBR seagrasses there are also species which possess different growth rates, having tissues which persist for different lengths of time. A wide range of characteristics have been measured in the monitoring of seagrasses in the GBR (Table 7).

Table 7. Response stages of seagrass meadows to external stressors and the various indicators applied through observations made *(in parentheses and italics)* on seagrass meadows in the GBR region (adapted from Waycott and McKenzie, 2010).

Healthy population (measure)	Primary sub-lethal stress (within plant)	State change (whole plant / meadow change)	Population decline	Population extinction
Tissue nutrient concentrations	− Evidence of light and/or nutrient limitation *(ratios of key macronutrients indicate light limitation at some sites and longer-term data indicate nutrient (N & P) loads increasing over time)* − Change in chlorophyll concentrations *(data suggest chlorophyll initially increases then rapidly declines when other metabolic demands outcompete)* − Change in storage of carbohydrates *(expect reductions, data not yet available)*	Threshold reached		Loss of seagrass
Production of reproductive structures (flowers and fruits)		− Reduced flowering and fruiting *(high variance in flowering and fruiting)* − Loss of seeds for meadow recovery *(high variance in seed banks)*	Threshold reached	
Change in plant morphology		− Reduction in area of photosynthetic tissue (e.g. smaller leaves) *(leaf width with evidence of low light – tissue C:N ratios)*	Threshold reached	
Community structure		− Change in species composition *(shifts in spp composition appear related to events, more analysis required)*	− Loss of species *(yet to identify key species changes with population decline)*	
Community structure (cont'd)		− Change in canopy height, epiphytes and algae *(highly variable more analysis needed)*		

Healthy population (measure)	Primary sub-lethal stress (within plant)	State change (whole plant / meadow change)	Population decline	Population extinction
Population structure (% cover, biomass, density)		− Change in abundance of species *(% cover change associated with variation in environmental parameters)*	− Reduction in number of genetic individuals within populations *(expect reduction under stress)*	
Meadow area			− Reduction in total area of meadow *(38% sites declined in meadow area over last 4 years)*	
Recovery time following disturbance	Limited or no change	Delayed recovery time	Potentially no recovery if threshold reached	

The application of these indicators in the GBR is discussed below.

Tissue nutrient ratios

Plants residing in nutrient poor waters show significantly higher C:N and/or C:P ratios than those from nutrient rich conditions (Atkinson and Smith, 1983). For seagrasses, N:P ratios in excess of 30 are considered to be evidence of P limitation and ratios less than 25-30 are considered to show N limitation (Atkinson and Smith, 1983; Duarte, 1990; Fourqurean *et al.*, 1992; Fourqurean and Cai, 2001). Also, under conditions of light limitation the C:N or C:P ratios will be lower indicating that there is an excess of nutrients compared to the usage in production of new fixed carbon through photosynthesis (Abal *et al.*, 1994; Fourqurean *et al.*, 1997; Longstaff *et al.*, 1999). It is for this reason that comparing deviations in the ratios of carbon, nitrogen and phosphorous (C:N:P) retained within plant tissue has been used extensively as an alternative means of evaluating the nutrient status of coastal waters (Duarte, 1990). Analysis of GBR datasets indicated the following:

- **N:P ratios:** A weak negative correlation was evident between overall seagrass cover and increasing tissue N:P. Trends for particular species were significant for the correlation between *Halodule uninervis* tissue N:P and abundance. *Halodule uninervis* is predominately found in coastal habitats, which are generally considered nitrogen limited. The increase in *Halodule uninervis* abundance (percent cover) is reflected in the tissue nitrogen.

- **C:N ratios:** Seagrass cover was significantly correlated with tissue C:N (all species pooled); the higher the tissue C:N, the higher the seagrass cover. Changing C:N ratios have been found in a number of experiments and field surveys to be related to light levels (Abal *et al.*, 1994; Grice *et al.*, 1996; Cabaço *et al.*, 2007; Collier *et al.*, 2009). Experiments on seagrasses in Queensland have suggested that at an atomic C:N ratio of less than 20, seagrass may suggest reduced light availability (Abal *et al.*, 1994; Grice *et al.*, 1996; Collier *et al.*, 2010a).

- **C:P ratios:** Total seagrass cover increased with increasing tissue C:P. The median seagrass tissue ratio of C:P is approximately 500 (Atkinson and Smith 1983), therefore deviation from this value is also likely to be indicative of some level of nutrient enriched (lower C:P) or nutrient limited conditions (higher C:P).

- In comparisons between sediment and nutrient status of seagrass meadows there was no significant correlation between mean concentrations of adsorbed phosphate or ammonium in seagrass meadows sediments each year and mean total abundance (biomass and percentage cover). There were no long-term trends among these parameters.

Change in chlorophyll concentrations

Limited data are available for assessing chlorophyll concentrations in seagrass leaves as a monitoring tool. Research into variation in chlorophyll content associated with light limitation and temperature elevation has been conducted through MTSRF (Waycott and McKenzie, 2010). Field observations of variability in total chlorophyll content at four sites in the Wet and Dry Tropics regions of the GBR reflect light availability (Collier *et al.*, 2010a). This monitoring data support the idea that chlorophyll content may be valuable as a tool for monitoring recent changes in light availability.

Experimental testing of plant responses to elevated temperatures showed a chlorophyll response at the 40°C treatment (Collier and Waycott, 2010). Total chlorophyll concentration was significantly affected by water temperature ($p < 0.01$) being lower in the 40°C treatment. Chlorophyll concentration was also significantly ($p < 0.001$) affected by species and was higher in *H. uninervis* and *T. hemprichii* than the other two species. Chlorophyll concentration was not tested at the end of the highest temperature treatment (43°C) as most leaves were dead within two days (Collier and Waycott, 2010).

Additional information on changing chlorophyll concentrations is a data gap and represents a good potential indicator of short-term change in seagrass primary response status for GBR seagrass meadows.

Meadow-scale production of reproductive structures (flowers, fruits and seeds)

The inclusion of surveys for sediment seed banks in the Seagrass Watch and reproductive structure surveys in MMP protocols has allowed a foundation data set for the production of flowers, fruits and seeds (reproductive structures) across coastal GBR seagrass meadows to be developed. Like other parameters measured for coastal seagrass communities throughout the GBR there is a high degree of variability in the production of seagrass flowers, fruits and seeds. Across the three years of sampling associated with the MMP, several sites produced virtually no flowers, fruits or seeds, while others have been particularly fecund during the sampling period, in particular sites around Townsville in the Burdekin region and also the Rodds Bay sites south of Gladstone. The five order of magnitude variability in per shoot reproductive effort was not statistically associated with any single environmental parameter although the compounded affect of light limitation, nutrient limitation and meadow recovery status may explain the trend observed. Total reproductive effort, estimated as the sum of all reproductive structures counted per site over all sampling periods (equivalent to total reproductive effort) is considered to be a useful indicator of seagrass health.

Change in plant morphology

Evidence that the species of seagrass have variable morphology under different conditions is considerable, even among the species that occur in the GBR (Abal *et al.*, 1994; Udy and Dennison, 1997; Longstaff and Dennison, 1999; Longstaff, 2003; Mellors, 2003; Mellors *et al.*, 2005; McMahon, 2005; Walsh, 2006; McKenzie and Unsworth, 2009). Research associated with the MTSRF Project 1.1.3[6] (Collier and Waycott, 2010; Collier *et al.*, 2010a, 2010b) provides additional experimental evidence of plant morphological change associated

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[6] http://www.rrrc.org.au/mtsrf/theme_1/project_1_1_3.html

with light limitation and elevated temperatures. All experimental evidence demonstrates that under more severe exposure to environmental stress such as limiting light (Walsh, 2006; Collier *et al.*, 2010a) or elevated temperature (Collier *et al.*, 2010b) that the production of leaves, in both number and leaf area, is reduced.

Field monitoring of plant morphology associated with multi parameter field monitoring activities (e.g. MMP) indicates that leaf width is variable. There was no direct relationship with leaf width and percent tissue carbon alone. There was a statistically positive correlation between the C:N ratio and leaf width where the relative carbon content was elevated compared to nitrogen giving larger leaves; *Halodule uninervis* and *Halophila ovalis* leaf widths increase with increasing tissue C:N. A positive relationship between leaf width and C:P ratio was also detected. Given that it is understood that leaf structures get smaller with lower light availability, the field observations support the presence of a synergistic relationship between leaf growth and available light and nutrients, where at relatively high light levels the plants are nutrient limited, switching to being light limited relative to nutrient availability. Therefore, leaf area could be a useful indicator of seagrass health in conjunction with other measures.

Changes in community and population structure

Significant amounts of data are available on the changes in species composition through Seagrass-Watch and MMP. These data are highly variable and no clear trends in species composition were determined at this time. However, the impact of local site 'events' such as storms, sediment movement, or other physical disturbance (e.g. Collier and Waycott, 2009) were not able to be incorporated into analyses at this time as information on these events was anecdotal. It is suggested that strong correlations between the loss of certain species in well established meadows may act as an indicator of change, however further evaluation of data incorporating best information on local events will be required to explore these trends more clearly. This represents a data gap at this time.

Changes in population structure are at present not being monitored directly, other than those associated with change in abundance and species composition. Analysis of the number of individuals within populations would be beneficial to understanding these trends (e.g. Collier and Waycott, 2009).

Change in seagrass meadow size

Meadow area, as measured by the MMP edge mapping activities, was able to be assessed for overall trajectories for the period 2005 to 2009 following the method applied in Waycott *et al.* (2009). Across all trajectories, the estimate for the rate of change is strongly negative although there was no statistical difference for trajectory with NRM region, habitat or overall trajectory. These trends are consistent with the dynamism of seagrass meadows in the GBR (sensu Collier and Waycott, 2009). However, the relatively large proportion of sites experiencing meadow decline (16 of 29 having negative trajectories, eleven of these significantly so) is of concern. This corresponds to the evaluation by McKenzie (2010) who estimated a large number of meadows in a relatively 'poor' state. Area of seagrass meadows is therefore considered to be a useful indicator of seagrass status.

Seasonal variability in meadow abundance measures (percent cover)

Measures of seagrass meadow status, canopy height, algal cover and epiphyte cover were highly variable. Epiphytes and algal cover showed no erratic seasonal or site variation. Canopy height varied seasonally however some sites showed taller canopy in the dry season, others the late wet season. A high level of variability in seasonal seagrass cover is detected across all sites monitored in the MMP and other Seagrass-Watch sites. It was more common that sites showed maximum abundance in the late monsoon season (sampling

usually in late March to early April) although some sites consistently showed the late dry season (October sampling) to be maximum cover. It is hypothesised that in sites where light is not limiting, the late monsoon season represents the time when temperatures are warmer, nutrients in abundance following summer flooding and terrestrial inputs are evident but not in excess as occurs during early monsoon 'events'. Other measures of abundance (biomass, density) are not regularly measured, in part due to the destructive sampling required and the additional labour required to assess these parameters. Some experimental projects include assessment of biomass (Waycott and McKenzie, 2010) however these data are from fewer sites and not considered further here.

Light

James Cook University researchers are exploring the relationship between light intensity and intertidal seagrass monitoring data, and the relationship between light and subtidal and intertidal seagrass responses in an effort to elucidate thresholds and the role of light as a driver in seagrass meadows (Waycott and Collier, 2009). Light loggers have been deployed at four intertidal and subtidal seagrass meadows in the Burdekin (Magnetic Island) and Wet Tropics NRM regions (Dunk Island, Green Island, Low Isles) since 2008. Total daily light reaching seagrass meadows at all locations was much higher in the intertidal meadows than the subtidal meadows. The annual pattern of total daily light at the seagrass meadow sites does not closely follow annual solar light intensity where light is highest around the summer solstice (December) and lowest at the winter solstice (June). Therefore, seagrass responses to light (e.g. percent cover) may also not follow annual changes in solar radiation.

Large peaks in light intensity occur during the winter period when spring low tides result in significant periods of time exposed to the air or very shallow water. This can result in a large departure in daily light between subtidal and intertidal seagrasses, particularly for intertidal sites above +0.8 m LAT, which is a typical low tide level throughout the year. Daily light was lowest at Low Isles and Magnetic Island, which relates to site depth and may also reflect a poorer water quality at these sites. However, light at canopy height data are not a true indication of water quality as the sites are at different depths (subtidal ranging from -0.9 to -2.5 m below LAT, intertidal from +0.6 to +0.9 m).

Further investigation of these findings will utilise water quality logger data (chlorophyll and turbidity loggers) to elucidate the source of changes in light at the seagrass meadows.

To conclude, the final list of indicators recommended for assessing GBR seagrass health are:

1. Tissue nutrient ratios (N:P, C:N, C:P);
2. Seed bank and reproductive structure surveys;
3. Leaf number per shoot and area;
4. Seagrass species composition; and
5. Changes in seagrass meadow cover and area.

Other indicators may permit greater inferences to be made on the responses of seagrasses to changing water quality however additional research will be required to evaluate their efficacy.

Estuaries

A number of studies have been conducted to develop and test appropriate approaches to the evaluation of estuarine ecosystem condition in a tropical Australian context (see Sheaves *et al.*, 2010; Sheaves and Johnston, 2010; Sheaves *et al.*, in press). Four categories of indices showed promise and each was evaluated for its utility to detect impacts and, for the more useful indicator variants, how suitable they were for reporting and communicating to technical and, particularly, non-technical end-users (Table 8). Additional studies further evaluated the two most promising indices (a) a suite of simple indicators based on fish assemblage composition (IS indicator set), and (b) carbon and nitrogen stable Isotope values.

With the exception of highly degraded sites, one feature that is consistent across most variable sets is the lack of clear faunal gradients that correlated with apparent impact gradients. Rather, assemblages showed strong among-estuary spatial and within-year temporal differences which probably related more to ecological process and estuary-specific habitat mixes than to impact status. This means that fish assemblage-based measures are unlikely to be effective at assessing estuary condition if assessment paradigms are based on the expectation of being able to define 'absolute' measures of condition. The fact that differences among estuaries are maintained from year to year means that a more profitable approach to assessing condition is to evaluate changes in fish assemblages over time relative to estuary-specific baselines.

The Indicator Suite (IS) proved reliable at detecting major impacts and defining differences among estuaries, however, questions remain about its ability to sensitively detect low level impacts. Apart from two highly degraded artificial estuarine lakes, it was quite difficult to find sites that showed evidence of impacts on fauna. This is reflected in the lack of correlation between presumed impacts and assemblages noted above, and is probably due at least in part to the resilience of estuarine biota, that are by their nature pre-adapted to deal with harsh and variable environmental conditions. Despite this, the IS indicator set provides a simple, effective and easily reportable approach for describing spatial and temporal changes in fish assemblage structure, making it a potentially valuable tool for monitoring studies.

Overall, IS indicators are likely to be most useful for detecting changes in overall assemblage condition. In contrast, stable isotope values proved definitive for detecting one specific type of environmental impact; the entry of organic pollutants, such as sewage wastes, into estuarine food webs with systems with high exposure to sewage pollution demonstrating consistently high nitrogen isotopic ratios.

Table 8. Summary of new indicators developed and evaluated for estuarine ecosystem condition. Note: PoE = Probability of Encounter; CPUE = Catch Per Unit Effort. Source: Sheaves *et al.* (2010).

Indicator Category	Specific Indicator	Success as Indicator	Reportability	Utility
Change in fish assemblage taxonomic structure	Multivariate composition	Excellent	Good for technical audience, difficult for non-technical	B
	DeltaPoE	Excellent proxy of multivariate composition	Excellent	A
Overall assemblage indices	Overall (mean) CPUE	Very variable due to non-systematic variation in catches of schooling species		C
	Overall (mean) PoE	Excellent overall measure of occurrence	Excellent as part of an indicator suite	A
	Composite diversity indices (eg. H)	Ambiguous combination of species richness and equitability		C
	Species richness (s)	Excellent simple measure of an obvious parameter; as long as idea of a 'pristine standard' is abandoned	Excellent as part of an indicator suite	A
	Equitability based on CPUE (J_{CPUE})	Useful but in some circumstances can be unpredictably variable		C
	Equitability based on PoE (J_{PoE})	Excellent simple measure of an obvious parameter	Excellent as part of an indicator suite	A
Recruitment of offshore juveniles	Recruit persistence	Has potential but too difficult and time consuming to produce		C
Trophic composition	Overall trophic composition	Varies in complex ways so difficult to interpret		C
	Trophic ratios	No consistent pattern		D
	Trophic group richness	Temporal patterns but no clear spatial differences		C
	Phytodetrivore dominance	No consistent pattern		D
SI profiles of key species	Fish Stable Isotope Analysis	Excellent for detecting organic pollutants in food webs	Excellent but may be too technical for direct reporting	B
Scavenging pressure	Scavenging pressure	Good potential but needs more extensive evaluation	Needs development	C

Fish assemblages

Fish assemblages are often seen as ideal targets for monitoring and reporting estuarine status (Ward *et al.*, 1998) because fish are relatively large, easy to identify, taxonomically well understood and familiar to the public through their use as food and as targets for recreational fishing (Harrison and Whitfield, 2004). Despite their attractiveness, indices based on fish assemblages are not widely used in monitoring and reporting (e.g. Forbes *et al.*, 2008; EHMP, 2008). Even where estuary fish assemblages have been included in reporting they have often proved to be poor at differentiating estuaries with different apparent levels of ecosystem condition (Moore *et al.*, 2007).

Recently a number of large-scale studies (e.g. Ley, 2005; Sheaves, 2006) have shed light on the spatio-temporal variability in Australia's Dry Tropics fish fauna. However, this improved understanding has highlighted the difficulties of using fish assemblages as tools for monitoring and evaluation. Studies of nine estuaries spanning 180 km (Sheaves, 2006) and 21 estuaries spanning 650 km (Sheaves and Johnston, 2010) of the coastline of tropical northern Australia show assemblages of adjacent estuaries are likely to be no more similar than those of estuaries hundreds of kilometres apart. There was also no simple monotonic relationship between measures such as species richness or catch per unit effort (CPUE) and impact status (Sheaves and Johnston, 2010). As a result, there is no simple way of determining what a 'normal' assemblage should look like, given our current level of understanding. Consequently, it is difficult to compare assemblages among estuaries, to determine if one estuary is more impacted than another, to define control sites, or define absolute standards of estuarine condition or condition based on fish assemblage structure. However, the presence of unique fish assemblages that remain distinct among seasons (Ley and Halliday, 2003) and years (Sheaves, 2006) indicates temporal consistency within individual estuaries suggesting that fish assemblage structure can provide a useful measure of estuary condition or health, as long as a dynamic, site-specific view is taken. In particular, indices based on how likely it is to encounter a particular species (probability of encounter [PoE]), rather than CPUE, show considerable promise (Sheaves and Johnston, 2010).

Although the PoE indices developed by Sheaves and Johnston (2010) have many operational benefits and performed well in comparisons with multivariate analyses, they need to be developed to a stage where they can be used as components of integrated indicator packages. In this context, simple summary measures of community structure, such as measures of diversity and indices of overall abundance, have distinct advantages because of the ease with which they can be translated into communication products such as report cards. Additionally, notwithstanding the difficulties with defining 'normal' assemblage structure, there is still an obvious need to provide guidelines for developing useful initial baselines (Newall *et al.*, 2006; Sheaves and Johnston, 2010). The relatively consistent values of mean PoE, S and J_{PoE} across the natural estuaries suggests that with a large enough base of sample estuaries from a region, defining 'provisional' baselines is possible. However, in comparing to provisional baselines it is critical to carefully consider goals and expectations. Baselines need to be tight enough so that indices would reliably provide early warning of situations where faunal composition was degraded. The corollary of this is that the indices should be expected to show some false positives; unimpacted estuaries that have naturally low numbers of species or low abundances. This should not be seen as a failure of the indices or the baselines but an indication of the sites that require more detailed investigation to determine why index values are anomalous. It is also important that provisional baselines are continually re-evaluated in the light of the ever increasing body of data accumulating for study estuaries as a result of monitoring.

No matter how potentially useful an indicator suite might be its value is ultimately determined by the quality of data on which it is based (Seegert, 2000). This means that careful sampling design and implementation are critical (Cooper *et al.*, 1994; Badenhausser *et al.*, 2007).

Seasonal variation in faunal composition means that the timing of sampling is critical. For tropical Australian estuaries faunal composition is consistent for low recruitment seasons so samples should be collected during the post-wet and dry seasons and sampling avoided during the late pre-wet and wet seasons. Variability in index values can be minimised by stratifying by habitat type. Here the interaction with the sampling gear used is important. Gear needs to be appropriate for the dominant habitat(s) available at an estuary, so they collect nekton efficiently to produce stable, repeatable data and represent the fauna of the estuary as completely as possible. Gear that can be used to collect larger numbers of replicate samples per unit time and that can be used across a variety of sites will usually be preferable, to allow faunal representation to be as spatially extensive as possible and to allow PoEs to be based on as many replicates as possible. Consistency of sampling is also important. So, while the measures described here are simple enough for community monitoring, quality outcomes require extensive operator capacity building and continual quality control (Seegert, 2000).

In conclusion, major impacts are detectable using fish assemblage based indices for impact assessment in tropical estuaries (Sheaves and Johnston, 2010; Sheaves *et al.*, in press) but the situation is not so clear for less extreme impacts. It seems possible that, because tropical estuarine fish are adapted to deal with harsh and variable environmental conditions (Kennish, 2002; Elliott and Quintino, 2007), they are very resilient to many sorts of impacts and so may be an inappropriate group for providing early warning of environmental degradation. There is clearly considerable work to be done to determine the extent to which tropical estuary fish fauna respond to low levels of impact before judgement can be made about their value as early warning indicators.

Carbon and nitrogen stable isotopes

One set of techniques with considerable promise in the detection of ecosystem impacts is stable isotope analysis (SIA) (Fry, 2006). SIA is particularly relevant to ecological assessment because isotopic ratios of common elements such as of carbon ($^{13}C{:}^{12}C$) and nitrogen ($^{15}N{:}^{14}N$) are directly interpretable in terms of food web structure. Carbon isotopic ratios (expressed in delta notation as $\delta^{13}C$) are recognised as indicative of sources of primary productivity, while nitrogen isotopic ratios (expressed in delta notation as $\delta^{15}N$) generally provide good indication of an organism's trophic level. Consequently, abnormal isotopic ratios can be interpreted as indications of trophic dysfunction. Several authors have reported that modifications to natural estuarine ecosystems have brought about changes in food web structure (Whitfield, 1996; Whitfield and Elliott, 2002; Powers *et al.*, 2005; Sosa-Lopez *et al.*, 2005) with a strong likelihood that change in structure was a response to shifts in primary productivity. From a reporting perspective, SIA provides a simple method of understanding the nature of productivity that underpins food webs.

SIA has some attractive features for long-term monitoring. For instance, stable isotope levels in animal muscle tissues represent an average measure of assimilated primary and secondary productivity over weeks to months (Fry, 2006), so outcomes are not as reliant on the timing of sampling as approaches based on catches of estuarine fish, which vary substantially over time, particularly in the tropics (Sheaves, 2006). A second advantage stems from known isotopic responses to particular environmental pollutants. For example, the presence of organic pollution tends to increase $\delta^{15}N$ levels, while the entry of nitrogen from artificial fertiliser sources tends to depress $\delta^{15}N$. However, although such effects are well known, it is also recognised that many other factors can influence the expression of such pollution signatures, meaning that the utility of SIA levels as indicators of condition needs to be trialled and evaluated before they can be validly used in a particular situation.

Trials of SIA in seven estuaries in three fish species in the Burdekin region showed clear differences in $\delta^{13}C$ signatures with generally consistent patterns of spatial differences for the

three species examined. However, shifts in carbon signatures did not correlate with differences in probable impact levels among locations. Results from further investigations at additional locations and across trophic groups supported these conclusions. Carbon values appear to be estuary-specific and driven by factors other than general potential for impact. Shifts in carbon values could not be directly attributed to possible agricultural, urban or industrial impact.

In contrast to δ^{13}C there was relatively little differentiation in δ^{15}N signatures among most estuaries. Although there was no gradient in nitrogen signatures across the potential impact gradient, it was evident that nitrogen signatures provided a strong indication of a particular type of impact, e.g. sewage discharge, providing clear evidence of elevated levels of organic nutrient (Costanzo et $al.$, 2001, 2003; Schlacher et $al.$, 2007).

Beyond the specific detection of sewage impact there was no clear indication that SIA would be useful as a comparative technique for directly assessing differences in 'general' estuary condition because both δ^{13}C and δ^{15}N values were not correlated with other probable impacts across the study locations. Carbon values differed among estuaries in an unpredictable way, suggesting estuary-specific differences in carbon sources that were not directly related to impact status. This aligns with previous work (Robertson and Duke, 1990; Ley, 2005; Sheaves and Johnston, 2010) showing strong estuary-to-estuary variation in the taxonomic composition of tropical estuaries. The δ^{15}N values showed only minor differences among locations that were not affected by sewage discharge.

The lack of differentiation in δ^{13}C and δ^{15}N values across estuaries with different probable urban and agricultural impacts suggests the technique has little value as an 'instant' measure of 'general' estuary condition that would allow the positioning of estuaries along a pristine-to-impacted condition gradient. This is not unexpected as it parallels the situation for the taxonomic composition of tropical estuarine fish assemblages where no simple pristine-to-impacted gradient is evident (Sheaves and Johnston, 2010).

From a purely scientific standpoint SIA thereby provides a rapid and simple method for assessing one component of estuary condition, however, whether the technique is suitable for incorporation into a suite of simple monitoring and assessment protocols that would be appropriate for use by managers and/or community groups is questionable. This is because of the relatively technical protocols associated with processing, analysis and interpretation of data, although the final presentation of data is interpretable enough to include in report card-type outputs suitable for general consumption.

Biomarkers

Biomarkers are particularly valuable as early warning signals of environmental degradation and provide an inexpensive, rapid and highly sensitive means of identifying and evaluating exposure to, and/or effects of, environmental contaminants in complex ecosystems. By selecting a key component in the ecosystem, in this case the top-level predator, barramundi (*Lates calcarifer*), and measuring multiple biomarkers, including measures of molecular, genetic and physiological impairment along with chemical analysis, the ecological relevance of environmental contaminants may be more readily elucidated and thus integrated into environmental management strategies (Brown et $al.$, 2004; Galloway et $al.$, 2004a, 2004b).

In order to assess whether chemical contaminants were impacting upon the sensitive ecosystems of the northern GBR, barramundi were sampled from estuaries of five separate river systems, which represent varying degrees of impact from anthropogenic activities (Humphrey et $al.$, 2007). A multibiomarker approach was used in conjunction with chemical analysis of water and sediment from the five systems to try and characterise the relationship

between anthropogenic contamination and response of resident biota in estuaries along the north Queensland coast.

Water, sediment and barramundi (*Lates calcarifer*) samples were collected from five North Queensland estuaries along a perceived pollution gradient in 2002. They were processed and analysed for trace organic contaminants such as polycyclic aromatic hydrocarbons, polychlorinated biphenyls, organochlorine and organophosphate insecticides and metals. In addition, the pollution induced responses of a suite of seven biochemical parameters (phase I biotransformation enzymes (eg. EROD, P450), fluorescent aromatic compounds, DNA damage, RNA:DNA ratio and neurotransmission enzymes) and two condition indices (condition factor and hepatosomatic index) were measured in barramundi. The resulting database was subjected to uni- and multi-variate analyses in order to determine the most suitable biomarkers to assess pollution in North Queensland estuaries and to classify the environmental quality of the sites. Principal components analysis on the biochemical markers revealed that EROD, EROD/P450, DNA damage and, to a lesser extent, cholinesterase activity and fluorescent aromatic compounds were found to be responsive to contaminants in the environment while cytochrome P450, condition factor and the hepatosomatic index were found to be less responsive biomarkers. Of particular significance was the ability of the cholinesterase activity assay to detect the presence of organophosphate insecticides, compounds that are notoriously difficult to detect in environmental samples analytically. Discriminant analysis was used to classify the pollution status of the various estuaries. It appears that the best discrimination between the various sites was obtained using discriminant analysis on the biomarkers; however, further analysis using water quality parameters and levels of organic contaminants in water and sediment produced a similar pattern as found with the biomarkers.

This was the first study to employ multiple biomarkers in a resident fish species in Queensland, and has demonstrated the utility of applying a multibiomarker approach in conjunction with traditional analysis of contaminants in providing valuable information in environmental risk assessment.

To conclude, the most suitable indicators of estuarine condition in the GBR are:

1. Ambient water quality characteristics;
2. Change in fish assemblage taxonomic structure (Probability of Encounter – PoE);
3. Fish assemblage indices – Overall mean PoE, Species richness, Equitability based on PoE;
4. Fish Stable Isotope Analysis (although this is expensive and may be costly for regular reporting); and
5. Fish Biomarker Analysis (although this is expensive and may be costly for regular reporting).

4.5.4 Conclusions

In addition to continued monitoring of the existing indicators in the MMP, a suite of indicators show significant potential for future application. These are summarised below. There are also several indicators that require further investigation. Future research needs are highlighted in Section 7.

- Water quality **bioindicators**:
  1. *Porites* brightness
  2. Macro-bioeroder density in massive living *Porites*
  3. Foram index
  4. Macroalgal cover
  5. *Acropora* cover
  6. *Acropora*/Hard coral cover ratio
  7. *Turbinaria* cover
  8. Hard coral juvenile density
  9. Hard coral juvenile richness
  10. Soft coral juvenile density
  11. Soft coral juvenile richness

- Seagrass health:
  1. Tissue nutrient ratios (N:P, C:N, C:P)
  2. Seed bank and reproductive structure surveys
  3. Leaf number and area
  4. Seagrass species composition
  5. Changes in seagrass meadow area

- Estuarine condition:
  1. Ambient water quality characteristics
  2. Change in fish assemblage taxonomic structure (Probability of Encounter - PoE)
  3. Fish assemblage indices - Overall mean PoE, Species richness, Equitability based on PoE
  4. Fish Stable Isotope Analysis (although this is expensive and may be costly for regular reporting)
  5. Fish Biomarker Analysis (although this is expensive and may be costly for regular reporting)

The following indicators show promise but require further investigation and validation:

- Dominance shifts in key microbial groups in biofilm communities are suggested as a useful candidate bioindicator of tropical coastal water quality.
- Assemblage composition of foraminifera, but not of corals, is a useful indicator of short-term (years) changes in environmental quality. Foraminiferal assemblages are effective bioindicators of turbidity/light regimes and organic enrichment of sediments on coral reefs. Fine-tuning of the FORAM index will require further studies of foraminiferal ecology, especially with regard to host–symbiont relationships.

- Changing chlorophyll concentrations in seagrass leaves represents a good potential indicator of short-term change in seagrass primary response status for GBR seagrass meadows.

- Changes in seagrass population structure are at present not being monitored directly, other than those associated with change in abundance and species composition. Analysis of the number of individuals within populations would be beneficial to understanding these trends (e.g. Collier and Waycott, 2009).

- Scavenging pressure and biomarkers in estuarine systems showed good potential but required further evaluation.

4.6 Indicators of socio-economic influences on water quality

Many of the manageable risks to water quality in the GBR are attributable to human activity. The drivers of these activities and the impediments to changing them are primarily social and economic. In the absence of understanding social and economic issues, policy and program development are likely to be less effective and efficient. The uptake of particular management practices across industries and regions in the GBR is one socio-economic indicator that is recognised as critical for GBR water quality management, and is described in Section 4.3 above. Other indicators of socio-economic influences are described below.

4.6.1 Socio-economic monitoring and evaluation framework

In October 2007, the Reef Water Quality Partnership coordinated a workshop of experts in the field of socio-economic research to develop a framework to obtain socio-economic information to support water quality management in the GBR. A number of MTSRF researchers involved in socio-economic research participated in the workshop. While the framework was not intended to be totally comprehensive and other information would be required, it does outline the key elements and requirements common to virtually all social and economic dimensions of water quality management.

The framework, which is represented in Figure 18, identifies the need for social and economic information in four key areas:

1. **Baseline assessment:** to understand the social and economic values associated with the GBR; the drivers of management practices that create risks to GBR resource condition and trend (RCT); and to understand the broad benefits and costs of actions to enhance RCT.
2. **Policy design** (including modification to existing policy), to ensure effective and efficient policies are developed that account for social and economic impediments to changing practices.
3. **Policy implementation** to achieve desired changes. It is assumed that the actions targeted under the policy implementation phase are developed based on science and will lead to changes in the resource condition trend. This is the area where the linkages with biophysical sciences are vital to result in on-ground actions.
4. **Monitoring and evaluation** of social and economic drivers and values to feedback into a comparison of the baseline assessment to measure change, and a feedback to inform policy design (adaptive management). Note: Key feedback loops are indicated by dashed lines.

Table 9 examines these components in greater detail and provides examples of potential indicators.
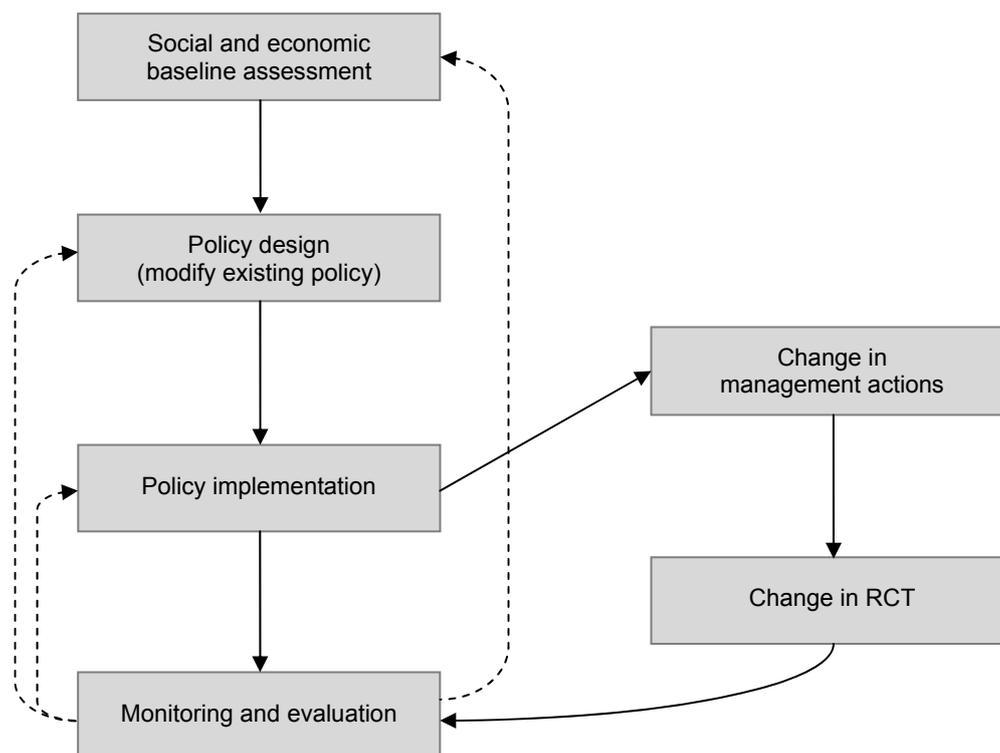
Figure 18. A social and economic framework for
water quality improvement in the GBR region.

Table 9. Potential indicators for a GBR water quality social and economic framework.

Information required	Indicator suite	Example indicators (primary = Pr; secondary = Se)
Baseline		
▪ Broad community makeup	▪ Baseline community indicators	▪ Population pressures (number/structure), labour market makeup, broad community resilience (Se)
▪ Economic trends, risks and market constraints	▪ Broad economic influences (power structures)	▪ Sector (e.g. sugar, tourism) trends, drivers and potential risks to/from change in water quality (Se)
▪ Tradeoffs between triple-bottom-line outcomes	▪ Individual aspirations ▪ Wellbeing and satisfaction	▪ Environmental, social and economic attitudes and aspirations relevant to enterprise management (Pr) ▪ Quality of life (Pr & Se)
▪ Capacity to change practice and resilience	▪ Capitals (human, social, financial, natural, built) ▪ Influences	▪ Qualifications, free financial capital, etc. (Pr & Se) ▪ General capacities and participation (Pr) ▪ Market, sector, social, policy regulation (Pr & Se)
▪ Likelihood of adoption to inform policy design	▪ Attributes and values of practices	▪ Public/ private benefit, compatibility, trialability, observability, acceptability, values, etc. (Pr & Se)

Information required	Indicator suite	Example indicators (primary = Pr; secondary = Se)
▪ Change in practice	▪ Characterise practices ▪ Risk to asset	▪ Identify practices, e.g. pesticide and fertiliser application rates, methods of application, stocking rates (S); several sources of data required ▪ Perceptions (P)
▪ Utilisation of GBR	▪ Use indicators	▪ Guest nights, reef visits, fishing, recreation, Indigenous use and interests, infrastructure and services (P & S)
Policy design tailored to requirements		
Rationale for policy and instrument choice	*Process monitoring* ▪ Socio-economic assessment of key policies, programs ▪ Desired outcomes / targets	*Examples* ▪ Utilising several indicators, undertake formal SEIA; information to inform prioritisation of what, who, where, how, when ▪ Program logic
▪ Likely ownership, responsibility and uptake of policy	▪ Engagement and communication (community and industry)	▪ Industries engaged, model adopted, involvement of regions ▪ Stakeholder analysis / network mapping
Policy implementation		
▪ Redesign of policy, was the policy implemented	▪ Governance and institutional ▪ Policy environment ▪ Funding resources ▪ Capacity ▪ Institutional arrangements ▪ Alignment ▪ Stakeholders	▪ Partners meet contractual obligations, assessment of institutional arrangements
Monitoring and Evaluation		
▪ Redesign of policy, informs update of baseline data ▪ Effectiveness of policy. ▪ Success / failure of policy intervention	▪ Monitoring baseline ▪ Evaluating change ▪ Monitoring and evaluating implementation ▪ Monitoring and evaluation design ▪ Uptake / compliance ▪ Outcomes	▪ Evaluate change in indicators to redesign policy; cost-benefit analysis ▪ Implementation process and content ▪ Percentage of target land holders utilising target management actions (P & S); Uptake / compliance ▪ Adoption of practice, continuation of practice (P & S) ▪ Expected outcomes

4.6.2 Management, governance and institutional indicators

The MTSRF has provided important insights into evaluation and reporting of water quality management responses (see Taylor and Robinson, 2010 for an overview of this work). Robinson and others have proposed an approach that addresses the integrated and diverse nature of water quality management programs across the GBR region, across various scales, and that recognises the growing support for monitoring and reporting approaches that enhance understanding and design of environmental management responses. Social science methods can be usefully applied for environmental management monitoring and evaluation because they help provide critical insights into and explanations of human behavior (see Robinson and Taylor, 2008). For example, Australia's Natural Heritage Trust Program has adopted a Monitoring, Evaluation, Reporting and Improvement (MERI) Program approach to assess program performance through the current status and trend in asset condition against immediate, intermediate, longer-term and aspirational targets. Here, the use of techniques such as 'performance story' reports that enable regional bodies to report on 'significant change' to regional assets and partnerships through their investment will provide complementary information to existing quantitative monitoring programs[7]. Internationally, there have been innovative efforts to develop participatory monitoring and reporting frameworks that fit the purpose of integrated project delivery and partner needs. Such applications draw on the literature regarding collaborative learning that emphasises the learning and communicative processes involved in interactions that expose conflict or build consensus rather than just the learning outcomes. Of particular interest to this framework are approaches that enable 'communities of practice' to engage in continuous assessment of progress achieved during program implementation. Developing a cross-regional framework to support adaptive governance of GBR water quality requires these factors to be considered. The dilemma faced with designing monitoring and reporting frameworks that consider different scales of activity and different logics or orientations in the mode of policy implementation is represented in Table 10.

Table 10. Logics and assertions at different scales of management.
Source: Taylor and Robinson, 2010.

	Science orientation	**Delivery orientation**	**Program orientation**
Policy level	Can the link between GBR level modeling and Water Quality Guidelines be strengthened?	Is the mix of regulation and voluntary instruments right?	Can we improve alignment of Nutrient Management Zones and WQIP investment and program logic?
Catchment or organisational level	How confident are we that the cause rather than symptoms are being targeted?	Did the system of stewardship payments result in accelerated uptake? Why, or why not?	Were funds allocated between and within catchments for maximum benefit?
Action / local level	How is local knowledge being incorporated?	Was the new technology effective? Were there unexpected impacts on-farm?	Outputs against milestones.

To address these issues the research team proposed and tested a cross-regional monitoring and reporting framework incorporating three functional components:

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[7] www.environmental.gov.au/nrm/MERI

1. **Conceptual modelling –** This requires the conceptual model of the planning and program logic to be explicitly described by participants, showing the main assumptions regarding water quality problems and their causes, and the pathways to influence.

2. **Sharing of action-outcome evidence across multiple catchments** – Through cross-regional networks, provide the forums for debate and exchange on each region's observations of delivery impact in the content of conceptual models they articulated in component 1.

3. **Negotiating collective responses from observations** – Based on revised conceptual models of impact / effectiveness of implementation activities from 1 and 2 above; roles, responsibilities and resources for the next round of delivery can then be negotiated. Importantly this cross-regional negotiation can incorporate the sharing, re-direction or refinement of implementation strategies that consider interdependencies between regions more explicitly and tensions or ambiguity between competing 'logics' or orientations of action, be they science, delivery or program based.

Application of this concept is further developed in Eberhard *et al.* (2008, 2009) through the preparation of an adaptive management procotol for GBR water quality management. This protocol aimed to assist water quality managers to explicitly state and test their water quality 'program logic' in terms of efficacy of management actions and contributions of partnerships in delivery of regional water quality plans. It includes steps to build and test knowledge used to inform water quality target setting, management activities and evaluation. It also supports improved accountability of stakeholders and investors. One component of the protocol was principles that reflected the policy and planning assumptions underlying adaptive management practice in GBR catchments.

The final phase of research related to governance arrangements for water quality management in the MTSRF involved the development and refinement of indicators for effective water quality governance (Robinson *et al.*, 2009b, 2010; Taylor and Robinson, 2010). In this context, good governance is the institutional capacity of regional body, government and industry actors to individually, and in partnership, promote knowledge integration. Table 11 presents the framework developed that uses knowledge indicators to evaluate collaborative governance performance. Results from testing this framework are presented in Robinson *et al.* (2009b). The approach enabled new insights, gained through application of the framework, to be incorporated into existing collaborative structures and decision-making processes. In addition, the timely feedback process helped to strengthen partnerships through providing a collectively developed agenda to guide deliberation and apply improvements to local arrangements. Specific improvements implemented by the regional body were focused around: regular communications with partners; enhancing partner engagement in planning and priority setting; and, further developing tools, processes, and procedures to support efficient and effective program delivery. This approach could be applied across all regional NRM groups in the GBR catchment, or elsewhere in national or international settings.

In addition to the governance and institutional investigations outlined above, a complementary component of management evaluation research in the MTSRF was designed to enable agencies, such as regional NRM bodies or state government agencies, to explore the likely impacts of interventions such as financial incentives or regulation to achieve water quality on the adaptive abilities of society (Lynam *et al.*, 2010a). Based on the outcomes of series of surveys with key stakeholders, a set of indicators of social resilience to environmental or other change in the GBR region are proposed, and included below in Table 12. A step-by-step user guide for applying this indicator framework, including survey templates and the most suitable assessment approaches, is provided in Lynam *et al.* (2010a), facilitating the application of this approach elsewhere.

Table 11. Knowledge attributes for collaborative water governance: an assessment framework applied to Reef Rescue grants delivery in the Wet Tropics NRM region. Source: Robinson *et al.* (2010).

Knowledge sharing functions	Attributes	Application context
1. Integration	**Diversity:** Multiple knowledge types (local, scientific, policy relevant) are identified and recognized. **Deliberation:** Institutions support debate between knowledge holders to frame problem and build understanding. **Inclusiveness:** Knowledge sharing and problem framing processes accessible and inclusive.	Scoping and problem/task framing stage. For policy development / resource allocation decisions.
2. Translation	**Credibility:** Knowledge used to inform priorities and actions is credible in terms of trustworthiness and adequacy. **Legitimacy:** Decisions and supporting knowledge legitimised through appropriate representation. **Salience:** The provision of knowledge and subsequent decision making is timely, and the type of knowledge is appropriate to problem context.	Design and implementation stage: To design policy implementation / resource prioritisation strategies.
3. Adaptation	**Relevance:** Measure of success or thresholds are cooperatively developed and are relevant to partners' views on 'good' implementation. **Roles** to monitor and evaluate impact and their respective domains agreed amongst partners. **Responsibilities** for sharing results from implementation, (i) between partners; and (ii) between scales of delivery, e.g. local-regional, are articulated. **Capacity:** Partners have the capacity to incorporate insights from review or monitoring into their own institutional behaviours.	Feedback for learning / assessing program effectiveness.
4. Impact	**Outputs and outcomes:** Monitors and reports on efficacy of partnerships to achieve water quality (and other negotiated) goals. May have short-term and long-term components and deliver social (i.e. building institutional capacity) and biophysical (i.e. improved water quality benefits).	Determines progress towards intended outcomes and positive / negative consequences.

Table 12. Current best set of indicators of social resilience to environmental or other change in the whole-of-GBR region. Source: Lynam *et al.* (2010a).

Current best list of indicators of social resilience	Broad category
1. Understanding of biophysical-social linkages 2. Biophysical system understanding 3. Social system understanding 4. Trusted and useable science 5. Access to technical experts or technical information 6. Long-term strategic decision making processes 7. Debate 8. Power to take action and affect change 9. Access to resources and incentives for experimentation 10. Collectively established goals 11. Stategic plan to achieve goals 12. Solutions or activities that achieved intended outcomes	**Group knowledge and capacity building**
13. Well-connected to social networks 14. Cross-scale networks 15. Organisational power 16. Orgaisational coordination 17. Congruence between government resource allocations (funding, support) and desired outcomes	**Networks and links to other organisations**
18. Economic wealth status 19. Econoically viable enterprises 20. Intact / healthy environment	**Other (economic and environmental)**

4.6.3 Conclusions

A set of indicators are proposed within a framework that recognises linkages between baseline assessment of primary social and economic indicators, policy design, policy implementation and monitoring and evaluation of key drivers (Figure 18; Table 9). While the indicators have not been fully tested, the framework provides a basis for further development of a program that could support monitoring and evaluation of key social and economic factors that influence water quality management in the GBR.

Several practical conclusions can be drawn from recent work on institutional and governance arrangements that have particular resonance for monitoring and reporting on the socio-economic and institutional dimensions of water quality management and broader governance in the GBR and elsewhere:

- Successful application of performance and assessment indicators requires both conceptual rigor – that is, grounded in social and institutional theory – and pragmatic agreement and testing of indicators amongst stakeholders. The participation of stakeholders in design, application of indicators and then interrogation of monitoring 'data' supports individual and organisational learning, contributing to adaptive management of water quality programs whilst in progress.

- Successful management at the regional scale is influenced by changes in the political, legal and social environment both above the regional scale, such as Federal-State relations or GBR-wide agreements – and below the regional scale, including the needs of local governments, individual farmers or catchment networks. Judging regional-level effectiveness must be considered within the vertical institutional connections that both enable and restrict regional action.

- In the case of diffuse water quality management, where substantive environmental outcomes are temporally and spatially distant, the use of indicators that evaluate the procedural success of planning and management activities provides stability, transparency and rigor to assessing management performance. It also contributes to building stakeholder trust and commitment to future cooperation and action.

The science that underpins the effective monitoring of factors that give rise to adaptation in social systems is new and much work still needs to be done. Through using a combination of qualitative enquiries and probabilistic modelling of qualitative data, significant progress has been made on developing, a) indicators of the likelihood of a social system adapting (Table 12); b) a process of collecting information on these indicators; and c) a robust approach to using this information to assess and predict the likelihood of adaptation.

5. Reporting results of monitoring and evaluation programs

5.1 Communication and reporting tools

Communication and reporting products are a critical component of any monitoring and evaluation program, and there have been substantial improvements in recent years on reporting techniques for water quality in the GBR. A review undertaken through the MTSRF in 2007 (Browne *et al.*, 2007) supported these improvements by providing a comprehensive review of relevant report card approaches from national and international report card programs, identification of some of the issues and requirements for developing a report card for the GBR region, and recommendation of principles and an approach on how to proceed with developing a report card for the GBR region. These findings are relevant to reporting in other national and international settings.

While there is no formal definition of a report card, in this document an integrated report card framework (IRCF) is defined as 'a scientifically valid approach for the integration and presentation of operational monitoring data in an accessible format for adaptive ecosystem management'. This definition captures the challenge of designing a report card to represent ecological condition that aims to statistically summarise a diverse range of data sources from a complex environment into a simplified form that remains scientifically valid. This definition also reflects the role that integrated report frameworks play within an adaptive management framework which involves ongoing monitoring and assessment. It also highlights the challenge of ensuring that the integrated components not only represent tools that managers can use, but are transparent to the scientific community and other stakeholders who require the detail underlying the report card grade. This places emphasis on the science behind the grade itself and its development. From review of the literature, a general framework for report card development is presented. The **general principles** resulting from the review, as well as the implications for the development of a GBR IRCF, are presented below with a note regarding the current status of each aspect for the GBR and how the MTSRF has contributed to the status. These principles can also be applied in other settings for the development of environmental reporting products.

Phase 1: Define the scope and objectives of the program, and establish what resources are available

> The scope of the IRCF must suit the available resources and time-frame. The goals and scope of the IRCF should be established as quickly as possible in order to guide subsequent stages. An overly ambitious or poorly defined scope will make it difficult to implement subsequent development stages. Existing data sets/monitoring programs may not be ideal given the requirements of the IRCF. Goals and scope of the IRCF may have to be revisited if they are to be constrained by available data.

> ***Current status:*** The Paddock to Reef Program now provides the basis for monitoring, evaluation and reporting of GBR water quality management (supporting the Reef Plan and Reef Rescue initiatives) and has clear objectives. The whole-of-system framework described in Section 4.1 (Bainbridge *et al.*, 2009a) and supported by the MTSRF provides the basis to the design of this monitoring program.

Phase 2: ***Understand the system, drawing together and documenting all relevant theoretical and empirical knowledge, as well as expert opinion***

Conceptual frameworks and conceptual models play an important role. Establishing the relationships between drivers, stressors and ecological impact is facilitated by explicit conceptual models of specific biophysical dynamics. An IRCF should encompass both the human and biophysical system, and provide clear direction in terms of management actions. Adherence to an appropriate and well-supported conceptual framework will assist here. An entire-system approach is particularly important, and the relationships between anthropogenic stressors, pollutant vectors, environmental conditions and ecological responses should be made explicit. Possible models include the traditional pressure-state-response approach as used in State of Environment reporting, the pressure-vector-condition used by the Queensland Government for its Stream and Estuary Assessment Program (SEAP), and the Millennium Ecosystem Assessment framework[8] that focuses on the linkages between ecosystem services and human wellbeing.

Current status: The first year of the MTSRF was focused on establishing agreed conceptual models for each component of its Water Quality Program. An overview of these is provided in the companion report, 'Catchment to Reef Connections' (Devlin and Waterhouse, in prep.), and was summarised in a baseline synthesis report and Year 1 summary of the MTSRF Water Quality Program (Brodie *et al.*, 2009d). This system understanding has supported the development and testing of the most suitable indicators for monitoring GBR water quality as described in Section 4.

Phase 3: ***Establish a measurement framework that will address the constructs defined in Phase 1 in terms of the systems identified in Phase 2***

This will include the definition of spatial reporting units (that should be scaleable), and the choice of indicators and their benchmarks. Consideration of hydrological dynamics and stream connectivity are important for determining the areas of human use that affect a given aquatic location and can form the basis for a multi-scale spatial classification. Regionalisation needs to consider the following:

- Latitudinal climate and landscape variation
- Definition and delineation of habitats
- Scales of variability (spatial and temporal) that occur within a patterned hierarchy of habitat and bioregion
- Identifying regions of minimal human disturbance that can be used to define appropriate benchmark reference conditions and thresholds-of-concern
- Establishing human disturbance gradients that can be used for validation of indicators, reference conditions, and regionalisation schemes.

Possible approaches to regionalisation may include:

- The use of landscape attributes or biological attributes singly or in combination.
- Clustering frameworks, with independent reference criteria for each cluster that may also be compared to model-based approaches. The influence of non-anthropogenic factors on indicators may be modelled and accounted for explicitly.

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[8] http://www.maweb.org/en/index.aspx

- Remote sensing methods can be useful in defining bioregions and mapping habitat extent.

- The use of expert opinion to help guide the selection of attributes and the identification of regional boundaries, especially for the reporting regions will be beneficial.

It is often possible to refer to national or international guidelines to determine reference levels for physical indicators (e.g. pertaining to water quality). Other indicators (biotic indicators in particular) will demand a process of reference level determination that is strongly related to the regionalisation scheme.

Current status: The whole of system framework described by Bainbridge *et al.* (2009a) which underpins the Paddock to Reef Program is based on a scaling approach (described in Section 4.1) across landscapes from the plot/paddock scale, to sub-catchment and catchment scale, to regional assessments and finally, GBR-wide scale. The multi-scale regionalisation developed for the GBR is a pragmatic compromise between the reporting units already determined through administrative arrangements, and the use of biophysical data of landscape attributes to determine a regionalisation capable of meeting a hierarchy of reporting needs. Investigation of the best approaches for reporting pollutant loads has been a focus of the MTSRF, outlined in Section 4.3. As described in Section 4, thresholds of concern have been defined for marine and freshwater ecosystems, and Water Quality Guidelines for the Great Barrier Reef Marine Park have been established (GBRMPA, 2009) supported by MTSRF research (De'ath and Fabricius, 2010 – see Section 4.5.1).

Phase 4: ***Establish an integration and reporting framework that will integrate and present the data generated in Phase 3 in a valid manner***

The organisation of the IRCF should be hierarchical, reporting at multiple levels of detail with regards to spatial, temporal and indicator specificity. This appears the ideal approach to accomplish the parallel goals of providing both transparency and methodological rigor.

The choice of how to integrate multiple indices can range from simple methods of averaging, or reporting the percent of sites and / or times an indicator meets a specified objective, through to more complex methods where individual indicators may be weighted, normalised to a common metric or interpolated over spatial or temporal reporting units. One of the major shortcomings of most of the methods reviewed is that they have no, or very crude, representations of uncertainty.

Recommendations about the strengths and limitations of each approach and the scope of works required for development and implementation are discussed more fully in subsequent reports. Internet presentation is recommended to maximise accessibility to the public. The use of active and interactive PDF technologies is of particular value.

Current status: Combining indicators to develop a metric or index has been one of the greatest challenges for scientists and managers to report GBR water quality status. However, monitoring providers for the MMP continue to work in collaboration with MTSRF researchers to progress this aspect of reporting, and the selection of robust indicators supported by the MTSRF provides a strong basis to progress this as a future priority.

Building on the outcomes of the above review, Kuhnert *et al.* (2007) proposed a conceptual and statistical framework for a GBR water quality report card. The study was undertaken with

the broad objective of developing a scientifically robust framework to support the production of report card(s) that integrate biophysical and socio-economic data from indicators that represent the pressures, vectors and responses in tropical aquatic landscapes of the GBR and Torres Strait regions. The project developed several outputs that could be utilised by interested organisations in the development of reporting products including:

- A description of the various phases required for indicator development and the associated statistical monitoring design approaches for data collection.
- Approaches for indicator assessment, integration and visualisation, including needs for data management and data sharing arrangements.
- A discussion of preferred statistical approaches that enable appropriate spatially focused reporting against targets or thresholds of concern.

General recommendations were also made for the further development of an Integrated Report Card for the GBR region. Many of these recommendations have now been adopted by management agencies in the GBR, and are being applied in the report card that is currently under development for the Paddock to Reef Program. Of particular interest has been the 'data wheel' proposed in Kuhnert *et al.* (2007) (see Figure 19) which was adapted and tested as part of the reporting approach for the MMP, but could be further progressed for the whole Paddock to Reef Program. Within the framework, there are four levels of data integration as shown in Figure 20:

1) Indicators;
2) Clusters (average of indicators);
3) Groups (average of clusters); and
4) Index (average of groups).

The three 'Groups' follow a Pressure-Vector-Response framework and represent Catchment, Freshwater and Marine zones. Within each of the groups are a set of 'Cluster' categories. The pressure or catchment indicator cluster categories are: (1) land use management and (2) land use type and condition. For the vector freshwater indicators, the cluster categories are: (1) physical, chemical and hydrological indicators; (2) ecology (bugs and fish); and (3) habitat condition. For the response or marine indicators, the cluster categories are: (1) nearshore and reef ecology; (2) marine water quality (see Figure 20). These cluster groups could be modified to more accurately reflect the recommended indicators in Section 4.

The data wheel approach was recommended for a number of reasons, including (i) it is flexible and new indicators, or indicator groups (e.g. socio-economic indicators) can be added at any stage; (ii) this type of visualisation option removes the need for different visualisation approaches for the scientific technical reports versus community report cards as different layers of the data wheel can be removed to reduce the complexity of visualisation where required; (iii) variations in the colour scheme can be used to demonstrate uncertainty in the data sets and potentially highlight where more data are needed for certain indicators; and (iv) this approach promotes consistency between the catchment, freshwater and marine approaches, rather than having different visualisation options as done for other report cards.

One of the outstanding requirements to support efficient and integrated reporting water quality in the GBR is the establishment of a shared data management system. The e-Atlas[9], established through the MTSRF, provides a means to share and access data, maps and information on topics relevant to the region, and could act as a suitable data repository for future GBR water quality reporting.

Figure 19. Example of GBR water quality data wheel approach that would be suitable for use in an integrated GBR report card. Source: Kuhnert *et al.* (2007).
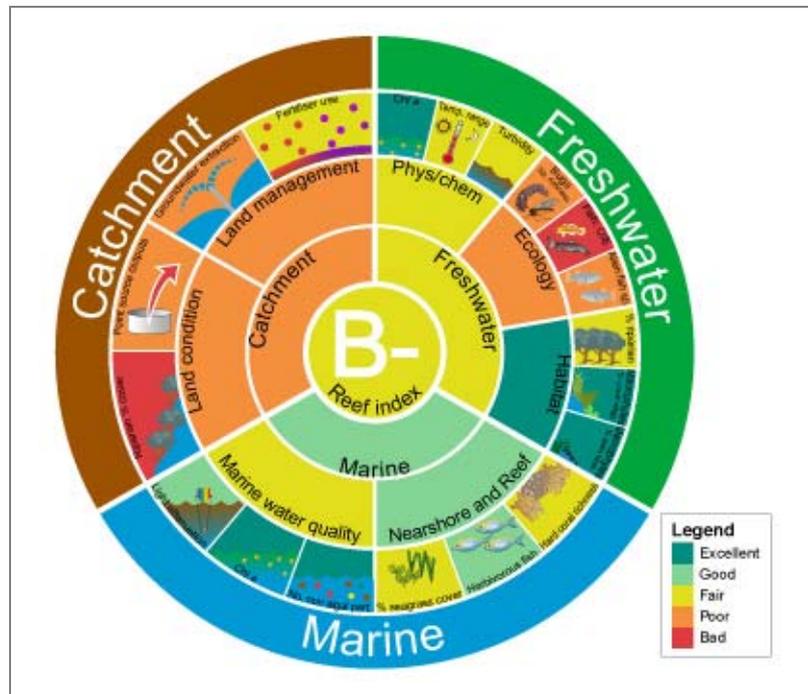
Figure 20. Example of the relationship used to build the index presented in the GBR water quality data wheel in Figure 19. Source: Kuhnert et al. (2007).

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6. Conclusions, management implications and future directions

6.1 Conclusions

The MTSRF has generated significant outcomes for informing the design and implementation of GBR water quality monitoring, evaluation and reporting programs. In particular, a monitoring and evaluation framework that incorporates biophysical, social and economic aspects of the system at multiple scales has been developed. This framework includes a range of monitoring and modelling activities to combine system attributes at several scales from plot/paddock, to sub-catchment, catchment and regional scales and, ultimately, across the entire GBR.

Suitable indicators for measuring ecosystem status and response have been developed and tested for the GBR and associated catchments. In some cases, thresholds for these indicators are established, which form the basis for the definition of guidelines to trigger a management response. The best ways to report indicators have also been considered.

The conclusions for each component described in this report are summarised in Figure 21 and presented below.

6.1.1 Management practices

Recording the adoption of defined classes or types of management practices is critical for measuring progress of land management improvement in the GBR catchments. Models can then be used to predict the water quality and economic outcomes of these improvements at various scales.

Financial motives are important in explaining adoption of management practices. However, there are other non-financial factors explaining the non-adoption of management practices by farmers. The presented research addresses this by incorporating heterogeneity among landholders and allows analysis of a range of policy scenarios to test the cost-effectiveness of management practice change and improving water quality in the region.

The following indicators are recommended for monitoring and evaluation of management practices for water quality management in the GBR:

1) Types / classes of management practices and rates of adoption;
2) Plot / paddock water quality runoff characteristics – sediments, nutrients and pesticides;
3) Cost effectiveness of management practices; and
4) Capacity to adopt new or improved practices through landholder profiling.

In addition, assessments that incorporate cost-benefit and water quality outcomes of particular management practices can assist in prioritising the adoption of practices and predicting outcomes of a set of management actions.

Management Practices

Management practices
- **Types of classes / management practices**
- **Adoption of practices (by class)**
- **Plot/paddock water quality – suspended sediments, nutrients, pesticides**
- **Cost effectiveness of practices**
- Capacity to adopt (landholder profiling)

Catchment health

Catchment health
- Habitat variables inc riparian extent & condition, flow, aquatic plants
- Physical condition inc velocity, bank stability, channel form, width, depth, sediment
- Water quality
- Invertebrate species & family richness
- Fish species richness & assemblage composition
- Abundance & diversity alien fish
- Floodplain lagoon & wetland benthic habitat
- **Wetland & riparian extent**

Catchment loads

Catchment loads
- **Sub-catchment & catchment water quality – suspended sediments, nutrients, pesticides**
- **Deterministic process-based model**
- **Statistical modelling framework - Loads Regression Estimator**

Socio economic influences

Socio economic influences
- Examples include:
- Population pressures
- Quality of life
- Public/private benefit of practice change
- **Use indicators e.g. No. guest nights, reef visits**
- General capacities & participation
- Evaluation of management performance
- Networks & links to other organisations

Estuarine health

Estuarine health
- **Ambient water quality – dissolved oxygen, pH, temperature, nutrients, suspended sediments**
- Change in fish assemblage taxonomic structure
- Fish assemblage indices
- Fish Stable Isotope Analysis ($$)
- Fish Biomarker Analysis ($$)

Marine Water Quality

Marine water quality
- **Ambient – suspended sediments, nutrients, carbon, salinity, temperature, chlorophyll, pesticides**
- **Flood plume – suspended sediments, nutrients, salinity, temp, chlorophyll, pesticides**
- **Flood plume extent & exposure**

Coral Health

Coral health
- *Porites* brightness
- Macro-bioeroder density in massive living *Porites*
- **Foram index**
- **Macroalgal cover**
- *Acropora* cover
- *Acropora* / Hard coral cover ratio
- **Hard coral juvenile density & richness**
- **Soft coral juvenile density & richness**

Seagrass Health

Seagrass health
- **Tissue nutrient ratios (N:P, C:N, C:P)**
- **Seed bank & reproductive structure surveys**
- **Leaf number per shoot & area**
- **Seagrass species composition**
- **Changes in seagrass cover & meadow area**

Figure 21. Summary of the recommended suite of indicators for monitoring water quality and ecosystem health in the Great Barrier Reef and its catchments, developed and tested through the MTSRF. Indicators highlighted in bold have already been adopted by the Reef Plan Paddock to Reef Integrated Monitoring, Modelling and Reporting Program.

6.1.2 Catchment and freshwater ecosystems

Previous work in the Wet Tropics has documented some thresholds for selected species and variables, including dissolved oxygen, nutrients, ammonia, substrate disturbance, sediment deposition and, more recently, dissolved oxygen. As supporting information to the development of thresholds of concern for freshwater ecosystems in the GBR, MTSRF research aimed to measure spatial and temporal variability of biophysical indicators in floodplain lagoons along natural environmental gradients and gradients of disturbance (see Pearson *et al.*, 2010a for an overview of this research). In particular, the field study in the Tully-Murray catchment was designed such that stressor-response relationships along gradients of disturbance (supported by data from laboratory trials and the literature) would help to identify thresholds – points along each disturbance gradient where ecological changes of scientific or management concern become apparent. The research showed that measuring such a threshold in the wetlands is challenging as disturbance gradients were difficult to interpret. Nevertheless, the results provided a robust set of stressor-response relationships and a substantial benchmark against which improvement in the ecological condition of floodplain lagoons can be evaluated.

A suite of recommended variables to describe/measure various freshwater habitats in the GBR catchments, resulting from the MTSRF research, are included below.

6.1.3 Streams

The Catchment to Reef Joint Research Program and MTSRF research on streams in the Wet Tropics (Pearson and Penridge, 1987; Pearson *et al.*, 2003; Arthington and Pearson, 2007) and in the Mackay-Whitsunday region (Clayton and Pearson, 1996; Leonard, 2009) has greatly informed our knowledge of how these ecosystems respond to human impact. Ecosystem health of streams could be monitored by measuring a suite of variables at multiple sites along natural stream gradients as follows:

- Habitat variables, such as flow regime, flow modification, stream geomorphic characteristics, riparian extent and condition (vegetation structure, weediness, canopy cover), aquatic vegetation and alien plant infestation, excessive algal growth, leaf litter, etc.
- Physical condition of the stream sites including: current velocity; bank stability; channel form; width; depth; sediment characteristics, including particle size and amount of detritus.
- Major water quality characteristics, including maximum and minimum values (measure through repeated 24-hr cycles) of temperature, conductivity, pH, dissolved oxygen, clarity, suspended solids, hardness, nutrients (mainly species of N and P) and short-, medium- and long-term variability in these metrics.
- Species richness of invertebrates ('species' here meaning taxa at highest level of resolution possible) and family richness of invertebrates – particularly good at the site/reach level.
- Fish species richness and assemblage composition – particularly good at the sub-catchment level.
- Abundance and diversity of alien fish species.

Note that following further investigation, aquatic plants were considered not very useful for monitoring stream health (apart from their habitat associations with the rest of the biota) because of their high level of variability (Mackay *et al.*, 2010).

Monitoring in contrasting seasons (late wet/early dry and late dry) is required to understand extremes of conditions.

Details of the preferred monitoring strategies can be found in the above references and via http://www.rrrc.org.au/catchment-to-reef/downloads/C2R-Arthington-Aet-al-2007-Biological-Indicators-Wet-Tropics-Streams.pdf

6.1.4 Floodplain lagoons

Previous research on floodplain lagoons (e.g. Pearson *et al.*, 2003; Perna and Burrows, 2005), more recent efforts by Pearson and others (see Pearson *et al.*, 2010a) in association with Wallace and others (investigating floodplain hydrology) have greatly informed our knowledge of the nature of these ecosystems and their biota, and how they respond to human impact. The research demonstrated that there are neither good reference (undisturbed) sites or highly impacted sites, in terms of aquatic biota, so very strong gradients of condition are not evident in the Tully-Murray lagoons. Higher levels of disturbance were evident in the lagoons in the Herbert and Burdekin systems (Pearson *et al.*, 2003; Perna and Burrows, 2005). Nevertheless, gradients in environmental variables and significant associations of the biota with them do exist across all these systems, so we are able to outline approaches to monitoring.

Ecosystem health can be monitored by measuring a suite of variables at multiple sites and times, with some exceptions, as follows:

* Habitat variables, such as flow regime, flow modification, lagoon geomorphic characteristics (including size and depth), aquatic vegetation and alien plant infestation, riparian extent and condition, leaf litter etc.
* Benthic habitat (plants vs. litter) and alien plant infestation were particularly important variables for invertebrates and fish, respectively
* Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P) and stratification; and short-, medium- and long-term variability in these metrics
* Invertebrate diversity (mainly family levels) and assemblage structure – provides a good benchmark with regard to habitat and water quality
* Fish species diversity and assemblage structure – provides a good benchmark with regard to habitat and water quality, and to connectivity and normal movements of fish
* Abundance and diversity of alien fish species
* Zooplankton assemblages were not very useful for monitoring because of their low diversity and the time involved in sample processing; however, presence or absence of zooplankton could be a useful and cost-effective measure in the event of severe deterioration of lagoon condition
* Monitoring in contrasting seasons (late wet/early dry and late dry) is required to understand extremes of conditions, including connectivity and success of dispersal/migratory activity.

6.1.5 Other wetland habitats

The MTSRF research was mainly restricted to the streams of the Wet Tropics and the lagoons of Wet Tropics floodplains, but previous research on floodplain lagoons in the Burdekin and Herbert systems (e.g. Pearson *et al.*, 2003; Perna and Burrows, 2005) in conjunction with results of Pearson *et al.* (2010a), allow comment on monitoring of floodplain lagoons across the GBR catchment. For *floodplain lagoons*, the suite of variables of utility in

monitoring is the same as indicated above for Wet Tropics lagoons. While the character of Dry and Wet Tropics systems differs greatly, differences are captured in the recommended suite of variables (including flow regime and temporal variation).

Riverine lagoons in the Dry Tropics (waterholes that remain when rivers cease to flow in the dry season) are the subject of two MTSRF-related PhD projects – one completed on water quality and algal dynamics (Preite, 2009), the other continuing on invertebrate dynamics and food webs (Blanchette, 2010). Results are not finalised but indications of metrics for ecosystem health monitoring are similar to those for Floodplain Lagoons and are as follows:

- Habitat variables, such as flow regime, flow modification, lagoon geomorphic characteristics (including size and depth), aquatic vegetation and alien plant infestation, riparian extent and condition, leaf litter, etc.
- Benthic habitat (edge, plants, sand, litter, riffle) and alien plant infestation are particularly important variables for invertebrates
- Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P), chlorophyll and stratification; and short-, medium- and long-term variability in these metrics
- Invertebrate diversity (mainly family levels) and assemblage structure – provide a good benchmark with regard to habitat and water quality
- Variability among lagoons and sub-catchments requires monitoring of multiple sites
- Algae are time-consuming to identify and show mixed signals with regard to ecosystem health, so are not currently useful for ecosystem monitoring.

Generally, the same variables will form the basis of monitoring programs of rivers and wetlands of different character, although the study designs will need to be modified to incorporate flow regime characteristics, and physical-chemical gradients in slow-flowing, intermittent and non-linear systems, such as floodplain lagoons. It also appears that, despite lack of active management of waterways and their surrounds for improved environmental outcomes, there is substantial resilience to impacts in those systems that receive good perennial flows.

The MTSRF Water Quality Program has also demonstrated how hydrological connectivity of floodplain wetlands can be quantified using hydrological and hydrodynamic modelling. This novel method can predict the timing and duration of connectivity of a large number of wetlands of different types under a range of flood sizes and can also be used to identify when water levels in a drainage network fall below critical thresholds for fish movement using readily available river gauge data. These types of relationship are central to the concept of setting environmentally acceptable flows, particularly where these are affected by significant abstractions for human use.

Quantitative connectivity modelling will also be useful to help explain the variations in wetland connectivity over time which may have important implications for (i) the movement and recruitment patterns of aquatic biota during and after flood events, (ii) wetland habitat characteristics and water quality, (iii) the biodiversity of individual wetlands over time, and (iv) the potential for wetland processes to influence the quality of water flowing to the GBR lagoon. As the hydro-dynamic model is driven by daily rainfall it should also be possible to quantify the potential impacts of climate change on wetland connectivity if the future changes in rainfall can be specified.

6.1.6 Pollutant loads

The recent development of baseline loads for the Paddock to Reef Program (see Kroon *et al.*, 2010) has highlighted several issues relating to the calculation of baseline pollutant load estimates for the GBR. At present, there are two primary approaches for estimating a pollutant load:

1. A (deterministic) process-based model (e.g. SedNet and the soon to be released WaterCAST) that incorporates mapped information about different sources of erosion and takes into account the hydrology and contaminant transport characteristics of the system. This information is used to route the pollutants through a river network and to estimate a load; and
2. A statistical modelling framework, LRE that makes use of monitoring data collected at a site within a catchment over a specified time frame.

The decision as to which model to use is largely subjective and depends on the resolution and representativeness of the data captured and how well the process model is believed to mimic the underlying hydrological processes and variability of the system. Where the monitoring data are representative of the river system, statistical approaches tend to be applied as in Kroon *et al.* (2010); when monitoring data are sparse or unavailable, process-based models are typically used. As a result, there is currently a mixture of the two types of models applied throughout the GBR catchments to estimate pollutant loads and inform a baseline in the Paddock to Reef Program (subjective analysis). In addition, process-based models are calibrated using monitoring data that are used as a means for calculating loads, ignoring uncertainty in the model structure as well as on the data that are used for calibration purposes. This mismatch of methods results in load estimates developed for different catchments, at different spatial scales, with different sources of error, making it difficult to monitor and track change (if any) through time and in space – an outcome which is considered a high priority for Reef Rescue Research and Development investment. To ensure that the resultant load estimates are beyond reproach, it is essential that all sources of uncertainty (parameter, model and data) associated with load estimates are propagated through the catchment models, resulting in transparent, objective and repeatable estimates of end-of-catchment loads.

The use of process-based models to estimate loads for paddock, catchment and marine components of the GBR is proposed in the design of the Paddock to Reef Program. The difficulty in relying solely on models (with parameters calibrated using monitoring data) is that monitoring data are unavailable in some parts of the GBR. This may result in an unrealistic and biased load estimate for these areas since the model is not calibrated to actual values. Further development of a model that assimilates both modelled and monitoring data is required to provide an objective and repeatable analysis for load estimation that accounts for the uncertainty in both the monitoring data and in the modelled estimates.

Another key aspect of pollutant load estimation from a management perspective is detecting changes over time, particularly in response to catchment initiatives designed to reduce loads. This is difficult because GBR pollutant loads can exhibit substantial inter-annual variability, driven by large variations in rainfall. In making the comparison between years we typically consider loads under 'average' conditions. This is what SedNet does as it is a long-term average. For the Paddock to Reef Program baseline pollutant load estimates (Kroon *et al.*, 2010), loads were averaged in some way over the different years that monitoring data were available. The LRE method currently provides load estimates for each year, drawing upon all available monitoring data to characterise relationships between flow and concentration. When data for a new year come in the model is updated and used to predict that year. This means it may take some time for new data to update the relationship enough to show up in

different load estimates. Modifications are, however, possible to improve the ability to detect changes over time (e.g. allowing time-varying relationships, comparing estimates derived from data over different time ranges). This may require an update to how 'average loads' are calculated if the focus is on detecting change rather than one-off baseline estimates.

The implications of the flood water quality studies in the Tully and Murray catchments, and potentially for other GBR catchments, are as follows:

1. Over-bank floods can make a large contribution to the marine load of sediment and nutrients and much of this load may not be recorded by standard river gauges.
2. In GBR catchments where floods are a significant proportion of the annual flow, current marine load estimates of sediment and nutrients (based on gauged flows, measured river concentrations and modelling) are probably too low, by significant amounts, depending on estimation method and constituent.
3. The size of this underestimate in any year will depend on the number and size of over-bank flood events in that year. This will make the monitoring of any underlying trends in ocean loads difficult unless it is possible to remove inter-annual variability.
4. Monitoring of marine loads will take a significant number of samples of both river and flood flows (in time and space) – otherwise the large uncertainties in mean loads may be misleading and it may be difficult to detect any load reduction trends.
5. The cause of the above underestimate in loads is mainly due to the poor recording of flood (over-bank) discharges by river gauges, but also to differences in flood water and river water quality concentrations.
6. Flood waters can carry more dissolved organic nitrogen (DON) than dissolved inorganic nitrogen (DIN) and this is the opposite of their concentrations in river water. Consequently DON loads to the ocean may be much higher than those previously estimated from riverine data.
7. Land management actions that focus on farm interventions in agriculture will potentially reduce DIN loads.
8. Reductions in DON (and sediment) loads that arise outside the floodplain require different interventions to those used in agriculture to reduce DIN; e.g. measures that slow down and reduce drainage and the introduction and/or rehabilitation of riparian zones and wetlands.

The inaugural flood water quality data collected in the Tully and Murray catchments has demonstrated the importance of obtaining observations from the key processes that control the marine loads of concern. In the Wet Tropical catchments studied, in addition to chanelised flow, over-bank flooding is a primary material transport mechanism and it is very difficult to adequately capture this process in monitoring and/or modelling schemes that are entirely river based. There is, therefore, a clear need to obtain estimates of the contribution that floods make to marine loads in other GBR catchments.

6.1.7 Estuarine and marine ecosystem health

Significant progress has been made on the establishment of thresholds of concern for coral reef and seagrass ecosystems. Abundances of a range of reef-associated organisms have been shown to change along water quality gradients. In addition, relationships between data sets of water quality, and macroalgal cover and the richness of hard corals and phototrophic and heterotrophic octocorals have been investigated at a GBR-wide scale (De'ath and Fabricius, 2008, 2010). This information has been used to define guideline values for water clarity and chlorophyll conditions, and provides the basis for the Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009). Further detail of these analyses is provided in Section 4.5.1.

The requirements for formation of healthy seagrass meadows are relatively clear as they are photosynthetic plants occupying a marine habitat. They require adequate light, nutrients, carbon dioxide, suitable substrate for anchoring, along with tolerable salinity, temperature and pH (Waycott and McKenzie, 2010). A number of thresholds of these requirements have been established for seagrass communities that are relevant to the GBR. The thresholds have been defined for tissue nutrient ratios that are related to light and nutrient availability. Other indicators are more variable and, to date, threshold values have not been established. Further evaluation of the best indicators of seagrass health and water quality conditions has been undertaken by Waycott and McKenzie (2010) and is summarised in Section 4.5.3.

In addition to continued monitoring of the existing indicators in the MMP, a suite of indicators show significant potential for future application for measuring reef ecosystem health. These are summarised below. There are also several indicators that require further investigation.

Water quality **bioindicators**:

1. *Porites* brightness
2. Macro-bioeroder density in massive living *Porites*
3. Foram index
4. Macroalgal cover
5. *Acropora* cover
6. *Acropora*/Hard coral cover ratio
7. *Turbinaria* cover
8. Hard coral juvenile density
9. Hard coral juvenile richness
10. Soft coral juvenile density
11. Soft coral juvenile richness

- Seagrass health:
  1. Tissue nutrient ratios (N:P, C:N, C:P)
  2. Seed bank and reproductive structure surveys
  3. Leaf number per shoot and area
  4. Seagrass species composition
  5. Changes in seagrass cover and meadow area

- Estuarine condition:
  1. Ambient water quality
  2. Change in fish assemblage taxonomic structure (Probability of Encounter – PoE)

3. Fish assemblage indices – overall mean PoE, species richness, equitability based on PoE

4. Fish Stable Isotope Analysis (although this is expensive and may be costly for regular reporting)

5. Fish Biomarker Analysis (although this is expensive and may be costly for regular reporting)

The following indicators show promise but require further investigation and validation.

- Dominance shifts in key microbial groups in biofilm communities are suggested as a useful candidate bioindicator of tropical coastal water quality.

- Assemblage composition of foraminifera, but not of corals, is a useful indicator of short-term (years) changes in environmental quality. Foraminiferal assemblages are effective bioindicators of turbidity/light regimes and organic enrichment of sediments on coral reefs. Fine-tuning of the FORAM index will require further studies of foraminiferal ecology, especially with regard to host–symbiont relationships.

- Changing chlorophyll concentrations in seagrass leaves represents a good potential indicator of short-term change in seagrass primary response status for GBR seagrass meadows.

- Changes in seagrass population structure are at present not being monitored directly other than those associated with change in abundance and species composition. Analysis of the number of individuals within populations would be beneficial to understanding these trends (e.g. Collier and Waycott, 2009).

- Scavenging pressure and biomarkers in estuarine systems showed good potential but required further evaluation.

6.1.8 Socio-economic influences

A set of indicators of social and economic influences on GBR water quality management are proposed within a framework that recognises linkages between baseline assessment of primary social and economic indicators, policy design, policy implementation and monitoring and evaluation of key drivers (Figure 18; Table 9). Several indicators suites are proposed and include monitoring of population pressures, environmental, social and economic attitudes and aspirations, quality of life, general capacities and participation, public/private benefit of practice change and use indicators such as guest nights and reef visits. While the indicators have not been fully tested, the framework provides a basis for further development of a program that could support monitoring and evaluation of key social and economic factors that influence water quality management in the GBR.

In addition, several practical conclusions can be drawn from recent work on institutional and governance arrangements that have particular resonance for monitoring and reporting on the socio-economic and institutional dimensions of water quality management and broader governance in the GBR and elsewhere. In particular:

- Successful application of performance and assessment indicators requires both conceptual rigor – that is, grounded in social and institutional theory – and pragmatic agreement and testing of indicators amongst stakeholders. The participation of stakeholders in design, application of indicators and then interrogation of monitoring 'data' supports individual and organisational learning, contributing to adaptive management of water quality programs whilst in progress.

- Successful management at the regional scale is influenced by changes in the political, legal and social environment both above the regional scale, such as Federal–State

relations or GBR-wide agreements – and below the regional scale, including the needs of local governments, individual farmers or catchment networks. Judging regional-level effectiveness must be considered within the vertical institutional connections that both enable and restrict regional action.

- In the case of diffuse water quality management where substantive environmental outcomes are temporally and spatially distant, the use of indicators that evaluate the procedural success of planning and management activities provides stability, transparency and rigor to assessing management performance. It also contributes to building stakeholder trust and commitment to future cooperation and action.

The science that underpins the effective monitoring of factors that give rise to adaptation in social systems is new and much work still needs to be done. Through using a combination of qualitative enquiries and probabilistic modelling of qualitative data, significant progress has been made on developing, (a) indicators of the likelihood of a social system adapting (Table 12); (b) a process of collecting information on these indicators; and (c) a robust approach to using this information to assess and predict the likelihood of adaptation. The indicators arising from this work can be grouped into four broad categories:

1. Group knowledge and capacity building (e.g. understanding of biophysical-social linkages, access to technical experts or technical information, collectively established goals).

2. Networks and links to other organisations (e.g. well-connected to social networks, organisational coordination).

3. Other (economic and environmental) (e.g. economic wealth status, economically viable enterprises, intact/healthy environment).

6.1.9 Communication and reporting

Communication and reporting products are a critical component of any monitoring and evaluation program, and there have been substantial improvements in recent years on reporting techniques for water quality in the GBR. A review undertaken through the MTSRF in 2007 (Browne *et al.*, 2007) supported these improvements by providing a comprehensive review of relevant report card approaches from national and international report card programs, identifying some of the issues and requirements for developing a report card for the GBR region, and recommending principles and an approach on how to proceed with developing a report card for the GBR region. These findings are relevant to reporting in other national and international settings, and were progressed into a recommended framework for a GBR water quality reporting product (Kuhnert *et al.*, 2007). The recommendations associated with the application of this framework warrant further consideration from management agencies in the GBR.

The MTSRF has generated significant outcomes for informing the design and implementation of GBR water quality monitoring, evaluation and reporting programs. In particular, a monitoring and evaluation framework that incorporates biophysical, social and economic aspects of the system at multiple scales has been developed. This framework includes a range of monitoring and modelling activities to combine system attributes at several scales from plot/paddock, to sub-catchment, catchment and regional scales and, ultimately, across the entire GBR.

Suitable indicators for measuring ecosystem status and response have been developed and tested for the GBR and associated catchments. In some cases, thresholds for these indicators are established, which form the basis for the definition of guidelines to trigger a management response. The best ways to report indicators have also been considered.

6.2 Management applications

The findings of this MTSRF research are directly relevant to managers of the GBR World Heritage Area and its catchment. Some examples of the management applications of the outcomes are provided below.

The multi-scale, multi-disciplinary 'paddock to reef' monitoring and modelling framework has been used to inform the development of the Paddock to Reef Program to support the evaluation of the Reef Plan and Reef Rescue initiative, and researchers continue to develop improved monitoring and evaluation techniques and indicators for continued refinement of program design. Many of the coral and seagrass indicators developed and tested through the program are already operational as part of the MMP.

The whole of system monitoring approach was also used in the development of regional water quality plans, including the WQIPs for the Tully, Barron, Townsville-Thuringowa (Black Ross), Mackay-Whitsunday and Burnett Mary regions. These programs assisted in the identification of priority contaminants and priority areas for each region. In conjunction with revised and improved pollutant load estimations, the findings have informed the prioritisation of Reef Rescue expenditure in these regions.

Research on thresholds of concern for coral ecosystems (De'ath and Fabricius, 2010) provides the basis for the Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009). These guidelines are used for assessment of the annual results of the MMP, for assessing annual status and relative change between monitoring periods. Remote sensing techniques that have evolved through development and testing in the MMP enable broad-scale assessment of chlorophyll, turbidity and Colour Dissolved Organic Matter concentrations against these guidelines. The thresholds that are suggested for tissue nutrients in seagrasses are currently used to assess the seagrass monitoring results in the same program.

The findings of the catchment and instream health research can be used to assess the condition of Wet Tropics streams and wetlands, which is of interest to the Queensland Government and regional Natural Resource Management groups. Where thresholds were not able to be established due to considerable local and regional differences and/or insufficient datasets, the assessment provides a robust set of stressor-response relationships and a substantial benchmark against which improvement in the ecological condition of streams and floodplain lagoons can be evaluated.

Finally, many of the most significant influences of the research on management decisions have been through the participation of MTSRF researchers in steering committees and technical groups coordinated by management agencies. MTSRF researchers are able to contribute their knowledge and synthesis of the research findings directly into the management processes; in many cases their contributions to discussion instigates interest which is subsequently supported through the provision of written evidence. Examples of these activities include the range of technical groups and forums coordinated for the regional WQIPs, design workshops for the Paddock to Reef Program and ongoing participation in the associated Technical Advisory Group, the expert workshops convened for the Multi-Criteria Analysis for prioritising Reef Rescue investment, participation in various committees for the Queensland Wetland Program, and involvement in several research prioritization workshops which have informed the Reef Plan and Reef Rescue R&D Strategies. Knowledge gained through the MTSRF and other research was also fed by participants involved in MTSRF research who wrote the 2008 Scientific Consensus Statement for Water Quality Management in the GBR (Brodie *et al.*, 2008a, 2008b) (with contributions from J. Brodie, K. Fabricius, R. Pearson, I. Gordon and J. Waterhouse).

6.3 Future research needs

The MTSRF research outlined above has also revealed knowledge gaps and new areas of research that should be progressed to inform continuous improvement of monitoring and evaluation programs, both in the GBR and elsewhere. The future research directions are summarised below for each system component that has been studied through the program.

Management practices

- Ongoing and proposed research building on the work already undertaken by van Grieken and others will assist in the estimation of the relative cost and effectiveness of improved management practices via better representation of the diversity of farm enterprises across land types and/or operating structures.

- Integrated research can greatly enhance and extend cost effectiveness estimates across diverse regions, including enterprise diversity and cost drivers such as physical constraints (e.g. soil types) and barriers to change (transition costs), thereby providing decision support to NRM regions in identifying the most cost effective mix of practice changes to invest in and to individual landholders to explore the cost and profit implications of BMP adoption for their farms.

Catchment and instream health

- Status and condition assessment for all GBR wetlands and waterways, and identification of priority wetlands in Queensland. There is a need for broader wetland monitoring for Queensland as, to date, only one major type of wetland (apart from streams) has been studied in relation to indicators of ecological status (i.e. the palustrine wetlands of the Tully-Murray floodplain, and one lacustrine wetland, Kyambul Lagoon).

- Wetland research needs to be extended geographically to further validate indicators of wetland health elsewhere within the Wet Tropics and into the Dry Tropics (e.g. the Burdekin River system). In addition, development of wetland monitoring protocols for other wetland types (only streams and lagoons included so far) is required for Queensland.

- Ecological condition of fish assemblages in floodplain wetlands given past wetland losses.

- Hydrological connectivity modelling could be useful for identifying better locations for an artificial wetlands where connectivity is considered important. This kind of connectivity modelling can also be used to identify when water levels in a drainage network fall below critical thresholds for fish movement using readily available river gauge data. These types of relationships are central to the concept of setting environmentally acceptable flows, particularly where these are affected by significant abstractions for human use.

- Quantitative connectivity modelling will also be useful to help explain the variations in wetland connectivity over time, which may have important implications for (i) the movement and recruitment patterns of aquatic biota during and after flood events, (ii) wetland habitat characteristics and water quality, (iii) the biodiversity of individual wetlands over time, and (iv) the potential for wetland processes to influence the quality of water flowing to the GBR lagoon. As the hydro-dynamic model is driven by daily rainfall it should also be possible to quantify the potential impacts of climate change on wetland connectivity, if the future changes in rainfall can be specified.

- Macroinvertebrate monitoring program design.

- Freshwater habitats in grazing lands – methods and framework for monitoring and management.

- Validating models in other Wet Tropics systems, in the Dry Tropics, and other contiguous systems (e.g. Cape York).

- Explicit tracking of fish movement within and across the floodplain to identify vital fish corridors between wetlands and the rivers, and across the agricultural landscape.
- Quantitative assessment of the degree of historical and recent habitat loss for fish in the GBR catchment, and the ecological condition of fish assemblages in response.

Pollutant loads

- Methods that integrate or fuse together models with monitoring data to facilitate robust estimation of pollutant loads with uncertainties in a repeatable framework.
- There is therefore a clear need to obtain estimates of the contribution that floods make to marine loads in other GBR catchments.

Estuarine and Marine Ecosystem Health

- Increased efforts to understand the linkages between system resilience to climate change and improved water quality.
- Understanding the linkage (synergistic/additive/dampening) between multiple stressors, in particular ocean acidification, temperature and nutrients. Recent work (Wagner *et al.*, 2010) shows that DIN enrichment enhances bleaching susceptibility in the Florida Keys. This confirms recent analysis/predictions from the GBR (Wooldridge and Done, 2009; Wooldridge, 2009). It is also known that DIN enrichment exacerbates the impact of increasing ocean acidification on coral growth (Renegar and Riegl, 2005).
- A continuation of biological and water quality inshore monitoring, incorporting water quality specific bioindicators (macroalgal abundances, hard coral and octocoral richness, coral recruit density and diversity, macrobioeroder densities in living massive Porites, the pigmentation or photosynthetic performance of corals, and the Foram Index). Consideration should be given to include Cape York into such a program.
- A comprehensive long-term river monitoring program remains essential to improve pressure and trend estimates in response to changing catchment management.
- Obtaining first-order approximations of local and regional residency times for dissolved and particulate materials. A receiving waters hydrodynamic model will improve estimates of flood plume dilution and dispersal, deposition and re-suspension of sediments, on biological and chemical transformations and help identify areas of greatest risk (e.g. exposure to highest loads, highest concentrations or greatest retention).
- Quantification of causes of intra- and inter-annual variability in concentrations of inshore nutrients and suspended solids (from floods, re-suspension, and Trichodesmium blooms). The spatial and temporal extent, frequency and duration of such extreme values needs to be quantified using long-term instrumental measurements of chlorophyll, benthic irradiance and turbidity samplers, and remote sensing.
- Improve understanding of cascading responses of habitats from sites of inputs to more pristine locations protected by distance from sources and their relative connectivity, i.e. estuarine aquatic plants and seagrasses–coastal mangrove and seagrasses–nearshore seagrasses and reefs–clearwater reefs.
- Compile data of light absorption and benthic irradiance in the GBR, to analyse its spatial and temporal distribution, its relationships to turbidity, Secchi depth, rates of sedimentation and hydrodynamic conditions.
- Investigate the biodiversity, ecological functions and water quality conditions of the Cape York inshore region as a matter of urgency, before climate change and other intensifying pressures start degrading this sole remaining Reference region that supports extensive coral reefs.

Socio-economic influences

- Further develop and test the proposed indicators of social and economic influences on water quality to establish a comprehensive monitoring and evaluation program for the GBR.

- Develop mechanisms to implement more effective uptake of management changes, including appropriate evaluation techniques.

Whole-of-system interactions

- Improved scientific coordination to synthesise and integrate data across disciplines to understand and quantify the linkages between catchment actions and the health of catchments and the GBR, and to assess whether catchment changes are sufficient to reverse water quality declines within a ten-year timeframe.

Progression of the future research directions highlighted above will assist managers of GBR water quality to improve the design of monitoring and evaluation programs within an adaptive framework. Continued alignment of monitoring programs with research programs, as has been the case for the MMP for several years, will assist in this process.

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